PORTLAND HARBOR RI/FS

APPENDIX E

REMEDIATION GOAL AND SEDIMENT MANAGEMENT AREA SENSITIVITY ANALYSIS

DRAFT FEASIBILITY STUDY

DRAFT

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This document is currently under review by US EPA and its federal, state, and tribal partners, and is subject to change in whole or in part.

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LIST OF ACRONYMS

µg/kg	micrograms per kilogram
AOPCs	Areas of Potential Concern
BaP	Benzo(a)pyrene
BaPEq	Benzo(a)pyrene Toxicity Equivalent
BERA	Baseline Ecological Risk Assessment
BHHRA	Baseline Human Health Risk Assessments
BTV	Background Threshold Value
BW	Body Weight
CERCLA	Comprehensive Environmental Response Compensation and Liability Act
COC	Contaminant of Concern
cPAHs	carcinogenic Polycyclic Aromatic Hydrocarbons
CV	Coefficient of Variation
DEQ	Oregon Department of Environmental Quality
DW	Dry Weight
EPA	U.S. Environmental Protection Agency
EPC	Exposure Point Concentration
FPM	Floating Percentile Model
FS	Feasibility Study
HQ	Hazard Quotient
IDW	Inverse Distance Weighting
KM	Kaplan-Meier
LOAEL	Lowest Observed Adverse Effect Level
LOE	Line of Evidence
LRM	Logistic Regression Model
LWG	Lower Willamette Group
mg/kg	milligrams per kilogram
MQ	Mean Quotient
ND	Non-Detect
NCP	National Contingency Plan
NN	Natural Neighbors
OC	Organic Carbon
PAHs	Polycyclic Aromatic Hydrocarbons
PCB	Polychlorinated Biphenyl
PDFs	Probability Distribution Functions
PEC	Probable Effect Concentrations
PEFs	Potency Equivalent Factors
PELs	Probable Effect Levels
pMax	Maximum probability of toxicity
PRGs	Preliminary Remediation Goal
QAPP	Quality Assurance Project Plan
RAL	Remedial Action Level
RAO	Remedial Action Objective
RG	Remediation Goal

RI	Remedial Investigation
RME	Reasonable Maximum Exposure
RMSE	Root Mean Square Error
RPD	Relative Percent Difference
Site	Portland Harbor Superfund Site
SMA	Sediment Management Area
SPI	Sediment Profile Imaging
SQG	Sediment Quality Guidelines
SQV	Sediment Quality Value
SWACs	Surface-area Weighted Average Concentration
TEQ	Toxic Equivalents
TZW	Transition Zone Water
UCL	Upper Confidence Limit
UPL	Upper Prediction Limit
UTL	Upper Tolerance Limit
WOE	Weight of Evidence

1.0 EXECUTIVE SUMMARY

1.1 PURPOSE

As part of the draft Feasibility Study (FS) for the Portland Harbor Superfund Site (Site), work was conducted to understand how various assumptions and calculations described in the baseline ecological risk assessment (BERA), baseline human health risk assessment (BHHRA), and site characterization sections of the remedial Investigation Report (RI; Integral et al. 2011) can be used to determine remediation goals (RGs) and sediment management areas (SMAs) to address contaminants that pose potentially unacceptable risk. This work was performed consistent with U.S. Environmental Protection Agency (EPA) sediment remediation guidance (EPA 2005), which advocates the importance of understanding the "sensitivity" of RG values proposed for contaminants of concern (COCs) to alternate assumptions about Site conditions and potentially unacceptable risks to ecological and human receptors.

As described in the risk management recommendations report prepared as part of the draft final RI (Kennedy/Jenks and Windward 2011), the results of the risk assessments together with information on background physical/chemical conditions in the Lower Willamette River, were intended to serve as the basis for establishing RGs and SMAs for COCs in the draft FS. The BHHRA and BERA identified the contaminants and the pathways whereby humans, fish, wildlife, and certain other ecological receptors could potentially be exposed at levels exceeding limits established by EPA and the Oregon Department of Environmental Quality (DEQ). The degree to which contaminants and exposure pathways posed potentially greater or lower concerns to ecological and human receptors was dependent on certain calculation assumptions and parameter values describing how those receptors could be exposed to contaminants. Alternate scientifically valid assumptions different from those required by EPA could have been used in the BERA and BHHRA. Using such valid alternate assumptions in the risk assessments would have resulted in different preliminary remediation goals (PRGs), and eventually different RGs, in the draft FS when compared to a single point estimate of an RG as set by the results of the risk assessments. Therefore, an analysis of how COC-specific RGs would vary for each COC-risk assessment pathway scenario and still result in values that are protective of human health and the environment is an important consideration when making final cleanup decisions (EPA 2005).

This section summarizes the results of the sensitivity analysis for RGs selected from the bounding COCs. The sensitivity analysis focused on sensitivities of polychlorinated biphenyls (PCBs) and benzo(a)pyrene toxicity equivalent (BaPEq) RGs (as well as comprehensive benthic risk areas). These two bounding COCs were selected as having the greatest potential impact on SMA identification.

1.2 APPROACH

With regard to addressing ecological issues in the draft FS, the *Risk Management Recommendations* report (Kennedy/Jenks and Windward 2011) identified the need to understand the protectiveness of PCB mink risks afforded by draft FS alternatives and to use river otter and bald eagle to analyze whether levels that are protective of potentially unacceptable PCB risks to mink are also protective of other ecological receptors (but not the benthic community, which is the focus of a separate assessment in the draft FS).

Based on those recommendations, this sensitivity analysis examined a broad range of total PCB PRGs. It analyzed the sensitivity of the PCB RGs to BERA assumptions about potential exposure, toxicity, and population-level risk. It calculated the probability of protectiveness of alternative PCB RGs for mink, and assessed whether mink RGs would be protective of other ecological receptors, based on analyses of river otter and bald eagle exposure assumptions.

The sensitivity analysis also focused on assumptions used in the PCB bioaccumulation model (Windward 2009b) to support both the BERA and BHHRA, because the results of the bioaccumulation model have a significant influence on RGs and SMAs, comparable to the influence of other assumptions that were considered in this sensitivity analysis.

With regard to addressing human health issues in the FS, the *Risk Management Recommendations* report (Kennedy/Jenks and Windward 2011) identified the need to examine several assumptions in the BHHRA addressing potential human exposure to PCBs and carcinogenic polycyclic aromatic hydrocarbons (cPAHs). Consequently, the sensitivity analysis examined exposure assumptions pertaining to human exposure to PCBs from consumption of smallmouth bass. The sensitivity analysis also considered human exposure to benzo(a)pyrene (BaP) from consumption of clams[1] and direct contact of Tribal fishers to cPAHs in in-water sediment. These three COC-risk assessment exposure pathway scenarios relate to select Focused PRGs.

In addition to risk assessment considerations, the procedures used to calculate background conditions and to map SMAs at the Site were examined. Because there are alternative scientifically valid methods and assumptions for the determination of 'background' COC conditions at the Site using upstream data, the sensitivity analysis explored how background estimates could vary to determine whether RGs might reasonably be considered to be at or below background levels. The factors examined in the sensitivity analysis included consideration of alternate upper-limit background statistics (e.g., upper prediction limit [UPL] and upper tolerance limit [UTL]), the use of dry weight or organic carbon (OC)-normalized data, treatment of outliers, reliance on a point estimate approach rather than a population comparison approach, and the handling of data below the detection limit for calculating total or summed values for different COCs. In addition, the sensitivity analysis examined estimates of central tendency in the background data sets describing different COC concentrations in sediment, including upper confidence limits on the mean and surface weighted averaging.

Because the SMA mapping procedures used in the draft FS to define spatial boundaries associated with active remediation areas involve assumptions that can be changed to

[1] Clam RGs were not developed due to uncertainty concerns discussed further in Section 3.3.

other valid assumptions, the sensitivity analysis also evaluated certain assumptions associated with the SMA mapping procedures. The assumptions included treatment of non-detect values in datasets, handling of data density artifacts and nearest neighbor contouring procedures, natural recovery of COC) concentrations over time, mapping of cPAHs in sediment, and the relationship between RGs and remedial action levels (RALs).

1.3 CONCLUSIONS

The results of the sensitivity analysis demonstrate that the range of potential PRGs and RGs for the Site extends from background to levels at baseline conditions in Portland Harbor. The results of the sensitivity analysis demonstrate that there is sufficient scientifically valid evidence that baseline conditions might already meet the Comprehensive Environmental Response Compensation and Liability Act (CERCLA) threshold criterion for overall protection of human health and the environment. The sensitivity analysis demonstrates that RGs above EPA's April 2010 Focused PRGs likely satisfy National Contingency Plan (NCP) protectiveness criterion. The sensitivity of RGs to uncertainties and assumptions about baseline risks and background conditions makes the NCP balancing criteria critically important in setting RGs and analyzing draft FS alternatives. Implementability and cost issues both become greater as RG values decrease.

With regard to ecological considerations, the sensitivity analysis concluded the following:

- The results of the sensitivity analysis indicate that the best estimate of the range of PCB RG values that are protective of mink potentially exposed to PCBs in sediment and meet EPA requirements for protection of the environment is from 79 to 640 µg/kg[2] (Figure ES-1). The mean estimate of the mink PCB RG from the sensitivity analysis is 256 µg/kg, significantly higher than the EPA point estimate of the RG of 31 µg/kg.
- The EPA point estimate of the mink RG of 31 µg/kg identified by EPA as protective of mink is lower than necessary to protect the three ecological receptors of most concern: mink, river otter, and bald eagle. The results of the sensitivity analysis indicate PCB RG values as high as 200 µg/kg will be protective of mink, river otter, and bald eagle.
- Benthic areas of concern were defined using guidelines established and provided by EPA on April 21, 2010 (EPA 2010). The principal uncertainty about these benthic areas of concern is whether EPA approves the implementation of these guidelines. These areas are called comprehensive benthic risk areas in the draft FS.

[2] All sediment concentrations presented in this report are on a dry weight basis unless otherwise noted. Sediment concentrations on other bases are always noted, as for example on an OC-normalized basis.

With regard to human health considerations, the sensitivity analysis concluded the following:

- The results of the sensitivity analysis indicate that the range of RG values for PCBs that are protective of human health from fish consumption (as represented by smallmouth bass) is from below EPA's Focused PRG background value 17 µg/kg to 6,346 µg/kg for 1 x 10^{-4} cancer risks (Figure ES-2 focuses on the lower end of the RG range as a comparison with EPA's point estimate of the smallmouth bass RG).
- The range of RG values for PCBs that are protective of human health from fish consumption based on noncancer endpoints is from below EPA's background value (17 µg/kg) to 373 µg/kg (Figure ES-3). The range of RG values for cPAHs that are protective of human health from direct contact with in-water sediment by Tribal fishers is 1,437 µg/kg to 3,702 µg/kg (1 x 10^{-6} cancer risk).
- The sensitivity analysis demonstrates that RGs higher than EPA's point estimate of the smallmouth bass RG of 29.5 µg/kg for total PCBs and point estimate of the sediment direct contact RG of 423 µg/kg for cPAHs may be protective of human health for both cancer and noncancer health effects.

With regard to the estimation of background values in sediment, the sensitivity analysis concluded:

- Alternate estimates of background concentrations of PCBs result in substantially higher and lower background levels than calculated using the single background point estimate of 17 µg/kg selected by EPA (Figure ES-4). The results of the sensitivity analysis indicate that, depending on the statistical method, background conditions for PCBs range between 5 and 37 µg/kg. The example for total PCBs relates to other COCs by showing that background is not one static point value, but rather a range of values against which alternatives can be evaluated.

With regard to SMA considerations, the sensitivity analysis concluded:

- SMA mapping procedures and calculation of RALs has an important additive impact on the overall spatial extent of SMAs at particular locations at the Site. There is a sensitivity range for RGs, as discussed above, that if mapped result in SMAs that range from designating the entire Site as meeting RGs that are protective of health and the environment, to identifying the entire Site as a one contiguous SMA. Though it is not realistic to evaluate a scenario with no SMAs (i.e., no action required), the results of the sensitivity analysis show that there is a range of scientifically valid SMA sizes that are larger or smaller than those generated through applying EPA assumptions.

In summary, the findings of this sensitivity analysis inform the evaluation of draft FS alternatives by providing a range of RG values that could be found to satisfy the CERCLA protectiveness criterion for the most widely distributed COCs (i.e., PCBs and cPAHs) and their associated receptors (i.e., adult, child and Tribal fishers and mink, river otter, bald eagle, and benthic community). This range is summarized in Table ES-1 The range of RG values aid risk management decision-making in selecting RALs that directly influence the spatial extent of SMAs in each alternative evaluated in the draft FS (see Appendices Da and Db of the draft FS). The range of RGs also informs decision-making in regard to determining how effective draft FS alternatives are likely to be at achieving protectiveness at various points in time after the remediation work is complete.

Table ES-1. Results of RG and Background Sensitivity Analyses

RG/PRG Evaluated				RG/PRG Basis	Range from Sensitivity Analysis
Total PCBs	29.5	µg/kg	EPA Point Estimate RG	Human health smallmouth Bass consumption, whole body, low ingestion rate, low bioaccumulation, 10^{-4} cancer risk	<17 - 6,346[3]
	31	µg/kg	RG	Ecological mink - diet	79 - 640[4]
	17	µg/kg	Focused PRG – Background	EPA UPL of upstream bedded sediment	5 - 37[5]
BapEq	423	mg/kg	RG	Human health direct contact with in-water sediment, 10^{-6} cancer risk	1,437 - 3,702[6]

[3] The range summarized represents the overall range of RGs calculated based on: whole body, fillet with and fillet without skin; 90, 95 and 99th percentiles of the risk distribution output for the target risk level of 1X10-4 for cancer risks; and the uncertainty in the bioaccumulation model as discussed in Section 3.3.

[4] The range summarized represents the overall range of RGs calculated via the "best estimate" of reduced kit production and bioaccumulation model, as discussed in Section 3.2.

[5] The range summarized represents the overall range of background estimates based on choice of statistic (other UPLs, UTLs, UCLs and percentiles); selection of outliers; substitution methods for calculation of totals; and, spatial autocorrelation as discussed in Section 4. It should be noted that the low estimate total PCB background of 5 µg/kg is based on a 95 UCL and not a UPL as is used for the EPA focused PRG background estimate.

[6] The range summarized represents the overall range of RGs calculated based on 90, 95 and 99th percentiles of the risk distribution output for the target risk level of 1X10-6 for cancer risk.

2.0 INTRODUCTION

Per U.S. Environmental Protection Agency (EPA) sediment remediation guidance,"(t)he remedy selection process for sediment sites should include a clear analysis of the uncertainties involved…The uncertainty of factors very important to the remedy decision should be quantified, so far as this is possible" (EPA 2005). Developing remediation goals (RGs) and mapping sediment management areas (SMAs) are key elements of the remedy selection process for sediment sites. Therefore, pursuant to EPA's guidance, this appendix presents an analysis of the sensitivity of sediment RGs and SMAs to changes in assumptions and exposure parameter values used in the baseline ecological and human health risk assessments (BERA and BHHRA, respectively), and also to changes in assumptions and methods used to evaluate physical and chemical conditions in sediments at the Portland Harbor Superfund Site (Site). A sensitivity analysis is consistent with EPA's sediment remediation guidance (EPA 2005) and contributes to the development of RGs and remedial action levels (RALs), which are discussed in Appendices Da and Db of the draft Feasibility Study (FS).

To support the draft FS, the focus of the sensitivity analysis presented in this document is to determine the range of RG values for bounding COC-risk assessment pathway scenarios identified in the BERA and BHHRA and to determine whether all or a portion of the different RG ranges satisfy the Comprehensive Environmental Response Compensation and Liability Act (CERCLA) threshold criterion (40 CFR §300.430(e)(9)) for protection of human health and the environment. This analysis focused on total polychlorinated biphenyls (PCBs) and benzo(a)pyrene (BaP) as two bounding COCs. Two other bounding COCs, dioxin/furan toxic equivalents (TEQ) and DDx, were identified in the risk management document (as summarized in Section 3.0 of the draft FS); however, the distribution of these other COCs are expected to contribute less to the overall impact on the draft FS alternatives evaluation. As described in the *Risk Management Recommendations* report (Kennedy/Jenks and Windward 2011), the results of the risk assessments, together with information on background physical and chemical conditions at the Site, were intended to serve as the basis for setting RGs and SMAs for COCs in the draft FS. The BHHRA and BERA identified COCs and exposure pathways whereby humans, fish, wildlife, and certain other ecological receptors could be exposed at levels exceeding acceptable limits established by EPA and the Oregon Department of Environmental Quality (DEQ). The degree to which COCs and exposure pathways posed greater or lower concerns to ecological and human receptors was dependent on certain calculation assumptions and parameter values describing how those receptors could be exposed to COCs. The sensitivity analyses described in this appendix concluded that several alternate scientifically valid assumptions different from those required by EPA could have been used in the BERA and BHHRA. Using alternate and equally valid assumptions in the risk assessments could have resulted in different preliminary remediation goals (PRGs) and eventually RGs. Therefore, the sensitivity analysis focused on examining how PCB and BaP RGs could vary for each risk assessment pathway scenario and still result in values that are protective of human health and the environment.

In addition to examining bounding COC-specific RGs, the sensitivity analysis also contributes to reaffirming the importance of the CERCLA balancing criteria in the draft FS alternatives analysis. Selecting the remedy alternative that represents the best overall risk reduction strategy (EPA 2005) for the Site given that the protectiveness criterion allows for a broad range of possibilities will require weighing the CERCLA criteria carefully because they counterbalance one another (EPA 1990). Balancing those criteria requires understanding the incremental costs and potentially unacceptable risk reduction associated with alternative RGs and remedial action objectives (RAOs) (EPA 1990, 1997; NRC 2001).

PRGs for human health and ecological receptors were provided by the Lower Willamette Group (LWG) to EPA in April 2009 (Windward et al. 2009a). Sediment PRGs were generated using risk assessment equations combined with target risk thresholds. For bioaccumulative-based sediment PRGs, target tissue levels were used with bioaccumulation models (Windward 2009b) to generate sediment PRGs. Sediment PRGs for benthic contaminants were based on Site-specific sediment quality values (SQVs).

Developing PRGs early in the RI/FS process based on readily available information is consistent with EPA guidance (EPA 1997, 2005), but guidance also points out that PRGs are subject to modification, based on consideration of uncertainties, among other things. Considering the effect of uncertainties on the PRGs is the focus of this appendix , thereby paving the way to the draft FS analysis of alternatives by defining the RG ranges over which the CERCLA balancing criteria could be applied.

In this sensitivity analysis, sediment RGs are discussed using the terminology in Section 3.5 of the draft FS and consistent with the rationale presented there for these terms. In summary, the term PRGs generally refers to goals that were developed at an early stage in the iterative process of remedial goal development and may be outdated or were not later refined into RGs, goals that are currently considered not sufficiently refined to be defined as RGs, and goals related to non-COCs (i.e., contaminants potentially posing unacceptable risk). Where goals are discussed in a way that spans all of these conditions, the term RG is generally used, since that is the primary focus of this sensitivity analysis and the draft FS. Additional discussion of the evolution of PRGs is provided in Section 3.0 of the draft FS.

The scope of the sensitivity analysis presented in this document addresses the following topics:

- Section 3: RG Sensitivity – This section describes changes to assumptions and exposure parameters associated with bounding COC-risk assessment pathway scenarios identified in the BERA and BHHRA, and the influence those changes have on both ecological and human health risk sediment RGs, including sensitivities associated with risk assessment uncertainties and assumptions and bioaccumulation modeling uncertainties and assumptions[7]. The sensitivity

[7] Bioaccumulation modeling is used to develop ecological and human health RGs for dietary exposure scenarios.

analysis identifies the range of RG values for the different COC-risk assessment pathway scenarios that satisfy the CERCLA threshold criterion for protection of human health and the environment.

- Section 4: Background Sensitivity – This section describes the sensitivities associated with developing background sediment values. This demonstrates the sensitivity of background estimates to uncertainty and assumptions about the upstream sediment dataset using scientifically rigorous and appropriate methodologies. This is important information when considering a background-based RG. The sensitivity analysis provides perspective on which parts of the Site are above background, which areas are not above background, and which areas remain uncertain with regard to exceeding or not exceeding background conditions (depending on assumptions used).
- Section 5: SMA Mapping Sensitivity – This section describes the sensitivity of SMA boundaries to uncertainties in the Site sediment data and assumptions in mapping procedures. The sensitivity analysis identifies the extent to which RALs ranging between the current Site surface-area weighted average concentrations (SWACs) to background levels could satisfy the CERCLA threshold criterion for protection of human health and the environment. The selection of RALs and SMAs (which are areas where sediment concentrations exceed RALs) becomes a risk management decision about how to balance the different CERCLA criteria to develop meaningful and feasible SMAs.
- Section 6: Summary and Conclusions – This section summarizes the findings about RG and SMA sensitivity to baseline risk assessment assumptions and uncertainties, and assumptions and uncertainties about physical and chemical conditions affecting the Site. The section provides overall conclusions about the implications of these findings for the draft FS. The findings will also be useful for identifying what is known and what critical parameters must yet be determined during pre-remedial engineering design studies.

3.0 REMEDIAL GOAL SENSITIVITY

This section addresses the sensitivities of the ecological and human health risk assessments and characterizes their influence on identifying potential ranges of RGs as compared to the single values (i.e., point estimates) of the RGs identified through the risk assessments (see Section 3.5 of the draft FS). Section 3.1 discusses bioaccumulation uncertainty, which affects both human health and ecological RGs. Sections 3.2 and 3.3 describe RG sensitivity to ecological and human health risk assessment assumptions and uncertainties. Attachments 1 and 2 contain additional technical details about the analyses presented in Section 3.

3.1 BIOACCUMULATION MODEL

Steady-state bioaccumulation models (Windward 2009b) were used to calculate sediment RGs from target tissue concentrations in fish (finfish and shellfish consumed by, for example, a person or mink). Bioaccumulation models can be used to predict fish tissue concentrations from sediment concentrations, or they can be "run backwards" to predict sediment concentrations from fish tissue concentrations. RGs are target sediment concentrations, so RGs are calculated from target tissue concentrations by running a bioaccumulation model backwards.

Total PCB RGs identified the most extensive SMAs for ecological and human health, so the sensitivity analysis focused on PCB bioaccumulation model uncertainty. There is uncertainty about PCB bioaccumulation in fish. The empirical data are uncertain because the exposure history of the fish that were caught and analyzed is unknown, as are the characteristics of the fish that were not caught. It is not known, for example, whether available fish catch information at the Site represents the average size or average age of fish in the population. It is also not known whether the fish not caught inhabited the same places as the fish that were caught. Because the empirical data are uncertain, simplifying assumptions are necessary in the exposure assessment and the bioaccumulation modeling.

Bioaccumulation model equations are uncertain too, because the mechanisms affecting uptake and bioaccumulation of COCs by fish are not completely understood (see for example McElroy et al. 2011) and the data that are needed to parameterize the model are incomplete and uncertain. Therefore, the model uses simplifying assumptions and parameter estimates. Because the bioaccumulation model contains uncertainty, a number that is put into the model produces an uncertain output. So, when a target tissue concentration is put into the bioaccumulation model, the sediment RG that the model produces contains the uncertainty inherent in the model. For example, the bioaccumulation uncertainty range on the mink PCB RG of 31 μg/kg is 20 to 40 μg/kg

(Windward 2009b).[8] The mink PCB RG of 31 µg/kg is considered to be the point estimate in the draft FS in comparison to the various ranges of RGs discussed here.

The impact of bioaccumulation model uncertainty, along with risk model sensitivity, in development of both ecological and human health RGs is discussed in the following two sections.

3.2 ECOLOGICAL RGs

3.2.1 Ecological RGs Associated with Risk Assessments for Wildlife Receptors

Based on the BERA (Windward 2011), PCBs are the primary contributor to potential wildlife risk in the Site, with mink, river otter, and bald eagle being at greater risk than other ecological receptors. The PRGs for these receptors were generated using risk assumptions that were intended to be much more likely to overestimate than underestimate risks. There are several assumptions that create uncertainty regarding what the appropriate range of RGs for these receptors should be.

In this section, we summarize RG sensitivity analyses that allow us to draw conclusions about the probability of protectiveness of an alternative range of total PCB RGs for all ecological receptors except the benthic community.[9] The sensitivity analysis demonstrates that there is a wide range of ecological PCB RGs that could satisfy the CERCLA protectiveness criterion for mink. The sensitivity analysis also evaluates whether RGs that are protective of mink are at least as protective of other ecological receptors.[10]

This information about the protectiveness of alternative RGs is summarized in Figure ES-1. The solid blue curve represents probability of protectiveness using best estimates of reduced kit productivity and PCB bioaccumulation. The dashed blue curves show the RG's sensitivity to uncertainty about reduced mink kit productivity and PCB bioaccumulation. The best estimate of the mean is indicated by the blue dot (256 microgram per kilogram [µg/kg]). The green dots bound the sensitivity range of the mean (129 to 477 µg/kg). The EPA point estimate of the mink PCB RG (31 µg/kg) is shown in orange. The red, white, and blue bands indicate level of protectiveness of other ecological receptors. The information on Figure ES-1 is discussed further in the remainder of this section.

[8] Steady-state PCB bioaccumulation model uncertainty was quantified in Section 5.6.7.2 of the bioaccumulation modeling report (Windward 2009). The example calculation of the mink-PCB PRG sensitivity to bioaccumulation model uncertainty was presented in Figure 5-35 of that report.

[9] The benthic community is covered separately in Section 3.2.2 because the BERA methods and conclusions are quite different than for all other ecological receptors. PCB RGs for other receptors are protective of the benthic community, but other aspects of benthic community risk need to be considered.

[10] Note that mink PRGs were never designated by EPA formally as Focused PRGs or RGs. However, EPA identified that its point estimate of the human health smallmouth bass Focused PRG could be considered as a surrogate for the EPA point estimate for the mink RG. Thus, the mink PRG can be considered as indirectly adopted into the Focused PRG list, and thus is termed as an RG in this document.

3.2.1.1 Mink Risk Assessment PCB RGs

A range of sediment PCB RGs was calculated considering a range of valid alternative assumptions about mink exposure, dose-response, and population-level effects (see Attachment 1-A for details). The changes are summarized in Table 3.2-1.

Table 3.2-1. Assumptions and Parameter Values Changed in the BERA pertaining to the PCB-Risk Assessment Pathway Scenario for Mink.

RG element	Sensitivity Analysis	Original PRG Model
Exposure analysis	Probabilistic exposure model considering variability and uncertainty in mink's diet	Deterministic exposure model
Habitat-use	Literature-based population model considering variability in Site habitat quality and variability and uncertainty in mink productivity, mortality, and dispersal	Equal use of all areas and no population dynamics
Effects analysis	Dose-response model with uncertainty bounds	Deterministic lowest observed adverse effect level (LOAEL)
Bioaccumulation model	Three most sensitive parameters (Kow, water temperature, and benthic invertebrate consumers' lipid content) were adjusted incrementally until the model output converged with the performance criteria	Parameter values selected to best estimate empirical fish tissue concentrations (maximize model performance)

The first step was to account for scientifically valid alternative assumptions about the mink diet. The PRGs were based on the deterministic dietary assumptions that were used in the draft BERA. The sensitivity analysis considered aquatic and terrestrial prey fractions reflective of mink diets reported in the literature and used previously in an exposure model developed by Moore et al. (1999). Aquatic prey fractions also accounted for the relative abundances of different fish species in the Site.

Dietary assumptions were quantified as probability distribution functions (PDFs) on prey fractions. The prey fraction PDFs were input into the Moore et al., model and combined with the calibrated PCB bioaccumulation model using Monte Carlo simulation. The Monte Carlo results provided a distribution on the total PCBs RG that could satisfy the CERCLA protectiveness criterion for mink. The resultant RGs ranged from 24 µg/kg (5^{th} percentile) to 217 µg/kg (95^{th} percentile) with a mean of 85 µg/kg. The model parameters contributing most to this range of RGs included mink's metabolic rate, the fraction of fish (especially carp) and mammals in the mink's diet, and the energy content of fish consumed as prey. So far though, this RG range only takes into account alternative dietary assumptions. It does not yet account for uncertainty about the mink's PCB effect threshold and therefore does not represent a complete range of potential RGs.

In addition to the probabilistic exposure and bioaccumulation models, two more models were needed to account for effect threshold uncertainty. The first of these is a stochastic population model, which looks at how mink survive and reproduce considering the extent and quality of Site habitat. The second is a model that quantifies the uncertainty about the PCB dose-response.

The stochastic population model looks at how mink would survive and reproduce in the Site habitat and predicts how many mink the Site could support. A model developed by MacDonald and Rushton (2003) for the Thames River in England was adapted to the Portland Harbor Site. The population model was sensitive to parameter assumptions such as the amount of habitat required per mink and natural mortality rates. The Site was found to be able to support from 0 to 23 mink using different assumptions.[11] A "stressor" was then added to the population model. The effect of the stressor was to reduce mink kit production (number of surviving kits per mated female). Simulations were run based on the mid-range of parameter values to determine the threshold for reduced kit production at which the Site's steady-state mink population began to drop. That threshold was found to be a 30 percent reduction in kit production, thus indicating that the population could suffer 30 percent mortality from an additional stressor (such as PCBs) and maintain a viable population. The dose-response model quantifies the uncertainty about the PCB dose that produces a 30 percent reduction in mink kit production. In contrast to the single lowest LOAEL used as the effects threshold used to develop the RG, this model reflects the variability and uncertainty in the relationship between PCB exposure and effects on mink fecundity (expressed as kit survival). Fuchsman et al. (2007) produced such a dose-response model, which was adapted using Portland Harbor data acceptability criteria. The Fuchsman et al (2007) model was used to translate the 30 percent reduction in mink kit production into an uncertainty distribution on the threshold dose giving a best (mean) estimate of 256 and a 90 percent confidence interval of 79 to 640 µg/kg body weight (BW)/day as compared to the BERA LOAEL of 37 µg/kg BW/day (Figure ES-1). By doing so, the Moore et al. exposure model and the Portland Harbor PCB bioaccumulation model were used to generate an RG uncertainty distribution that accounts for uncertainties in bioaccumulation, diet, dose-response, and population level effects.

After incorporating alternative assumptions about population-level effects and dose-response, the RG now ranges from 51 µg/kg to 919 µg/kg with a mean RG estimate of 256 µg/kg.

However, the analysis has not yet accounted for bioaccumulation uncertainty, which was described in Section 5.6.7.2 of the Portland Harbor RI/FS Bioaccumulation Modeling Report (Windward 2009b). Bioaccumulation uncertainty expands the 5th to 95th percentile range estimate for the mink PCB RGs by about 33 percent.

[11] Based on a survey of potential mink and river otter habitat conducted by a team of USFWS biologists, USFWS concluded that the lower 15 miles of the Willamette River is unlikely to support self-sustaining mink or river otter populations due to habitat constraints, though habitat on Sauvie Island and Multnomah Channel and nearby areas outside the Site (notably Ross Island and Oaks Bottom wildlife refuge) could support some individual mink, while some habitat within the Site could support individual dispersing juvenile male otters (USFWS 2011).

The cumulative effect on the mink RG estimates of scientifically valid alternative assumptions for all of the aforementioned factors is presented in Table 3.2-2 and Figure ES-1. The mean PCB RG for mink, based on best estimates of reduced kit productivity and PCB bioaccumulation, is 256 µg/kg dry weight (DW). The 5 to 95th percentile range is 79 to 640 µg/kg. Bounding assumptions for reduced kit productivity and bioaccumulation then extend the RG range by about a factor of two in either direction, indicating that the RG could possibly go as low as 36 µg/kg or as high as 1,192 µg/kg. EPA's preferred point estimate of the RG (31 µg/kg) falls below the lower end of this range.

Table 3.2-2. Total PCB RG Estimates for Mink Associated with a 30% Reduction in Kit Production (LCL, best estimate, and UCL), Including Exposure Uncertainty and Bioaccumulation Model Uncertainty

Type of Revised RG Estimate	Total PCB Concentration (µg/kg)		
	LCL on Reduced Kit Production and Lower Bound Bioaccumulation Model	Best Estimate of Reduced Kit Production and Bioaccumulation Model	UCL on Reduced Kit Production and Upper Bound Bioaccumulation Model
Mean	129	256	477
5th Percentile	36	79	148
Median	96	194	362
95th Percentile	330	640	1192

LCL – lower confidence limit
UCL – upper confidence limit

3.2.1.2 Protectiveness of Other Ecological Receptors

The uncertainty analysis for mink shows that the point estimate mink PCB RG, which is based on the PRG of 31 µg/kg, is lower than necessary to protect the mink population. The results of the sensitivity analysis applied to the mink-PCB risk assessment pathway scenario showed there is a wide range of PCB RGs that satisfy the CERCLA protectiveness criterion for mink. The point estimate of 31 µg/kg is highly conservative, and corresponds to a 99.9 percent probability of protectiveness, based on best estimates of reduced kit productivity and PCB bioaccumulation (Figure ES-1). This point estimate corresponds to a greater than 95 percent probability of protectiveness based on the lower-bounds of kit production and PCB bioaccumulation (the most conservative scenario).

This raises the question of whether a higher mink RG would still be protective of other ecological receptors. Bald eagle and river otter exposure assumptions were evaluated to address that question (see Attachment 1-B for details). As discussed in the *Risk Management Recommendations* report addressing ecological considerations

(Kennedy/Jenks and Windward 2011), mink, bald eagle, and river otter can be used to assess protectiveness for all non-benthic ecological receptors.

PRG estimates for bald eagle and river otter, expressed as percentages of equivalent mink RGs,[12] are presented in Table 3.2-3. For example, the mean PRG for bald eagle is nearly 9 times higher than the mean RG for mink (859 percent). The bald eagle PRGs are consistently higher (6 to 15 times higher) than the mink RGs, indicating that remedies protective of mink are also protective of bald eagle.

The river otter case is somewhat more complicated. Mink RGs up to slightly higher than the median estimate from the sensitivity analysis are at least as protective of river otter as they are of mink. Using best estimates of reduced kit production and bioaccumulation (the middle column in Table 3.2-2), 200 µg/kg is protective of both river otter and mink. This is represented by the red band in Figure ES-1. On the other hand, goals greater than 41 percent of the 95th percentile mink RG are not protective of river otter. The 95th percentile mink PCB RG, based on best estimates of reduced kit production and bioaccumulation, is 640 µg/kg (Table 3.2-2). Using best estimates of reduced kit production and bioaccumulation, RGs greater than 260 µg/kg[13] are not protective of river otter. This is represented by the blue band in Figure ES-1. RGs between 200 and 260 µg/kg (the white band in Figure ES-1) are protective of river otter, but less so than they are of mink.

Table 3.2-3. Factor Difference Between Revised PRG Estimates for Bald Eagle and River Otter Relative to those for Mink RGs[a]

	Bald Eagle	River Otter
Mean	9X higher	1.3X lower
5th percentile	16X higher	2X higher
Median	10X higher	1.0 (equal to mink)
95th percentile	6.2X higher	2.4X lower

a Mink RGs are presented in Table 3.2-2. The PRGs for bald eagle and river otter can be obtained by multiplying the mink RG by the factors higher, or dividing by the factor lower.

The main reason for differences in the PCB goal values for river otter and mink is because river otters have less varied diets than mink. The river otter is an obligate piscivore and derives most of its dietary needs from fish, whereas mink also prey on terrestrial organisms like birds and rodents. The upper end of the PCB RG range for mink reflects a greater reliance on terrestrial prey, an exposure scenario that does not apply to river otter.

12 For this analysis we only considered dietary uncertainties because in the case of river otter, mink is used as a surrogate for the PCB effect threshold (so the uncertainties effectively cancel out). For bald eagle we had insufficient data to develop a distribution on the effect threshold.

13 0.41 * 640 = 260

3.2.2 Uncertainty Affecting FS Analysis of Alternatives for Benthic Receptors

Benthic areas of concern, now termed "Comprehensive Benthic Risk Areas" for the draft FS, were defined in the risk management recommendations report (Kennedy/Jenks and Windward 2011) using a weight-of-evidence (WOE) framework called the comprehensive benthic approach. The comprehensive benthic approach is a risk management tool that was developed through a long, collaborative process based on a set of guidelines provided by EPA (2010).

The LWG and EPA have been working on the comprehensive benthic approach since early 2010, starting with a set of guidelines EPA gave the LWG in an April 21, 2010 letter (EPA 2010). The guidelines described EPA's goals for the draft FS analysis of alternatives for benthic assessment endpoints:

- Define areas that pose unacceptable risk to the benthic community
- Define the areas and volume of contamination that may pose risk to the benthic community
- Evaluate remedial action alternatives and effectiveness (did it meet the RAO)

The April 21, 2010 EPA letter also provided guidelines for evaluating remedy effectiveness:

- All benthic sediment quality guidelines (SQGs) in the March 24, 2010 list will be included in the analysis. If specific SQGs are found to be inconsistent with other lines of evidence (LOEs) listed below, EPA will review the analysis and determine whether these should be included in the draft FS.[14]
- Sediment toxicity bioassays will form the primary LOE for this analysis. The sediment toxicity LOE will include level 2 (moderate) and level 3 (severe) effects

[14] EPA used the pooled Floating Percentile Model (FPM) level 3 SQVs from the draft BERA as focused PRGs, and added PELs for arsenic, chromium, lead, nickel, zinc, and lindane (gamma HCH). EPA also added the PEC for copper even though the FPM SQVs already included a copper PRG, the PEC for total polycyclic aromatic hydrocarbons (PAHs) even though the FPM SQVs already included a total LPAHs PRG, and the PECs for DDD, DDE, and DDT even though the FPM SQVs already included a total DDx PRG. All of the benthic PRGs were noted as being subject to further evaluation in the comprehensive benthic approach (EPA 2010). Per agreement with EPA, the comprehensive benthic approach is to be applied in the draft FS, not the BERA (see item number 11 in EPA Clarifications to Resolution of EPA September 27, 2010 Comments on Benthic Risk Evaluation (submitted as Attachment B to the LWG's January 12, 2011 letter) (Humphrey 2011). The SQVs have subsequently been revised based on additional modeling and negotiations between the LWG and EPA, as documented in Attachment B to the January 12, 2011, LWG letter to EPA (LWG 2011a), the attachment to a February 25, 2011, RI/FS schedule letter from EPA to the LWG (Humphrey 2011), and the LWG's March 9, 2011, draft response (LWG 2011b) to EPA's February 25, 2011, letter. SQVs are now based on four individual FPMs and one pooled logistic regression model (LRM), each predicting two levels of toxicity (Level 2 [L2]—moderate toxicity and Level 3 [L3]—high toxicity).

for all endpoints (*chironomus* [sic] biomass and mortality and *hyalella* [sic] biomass and mortality).

- The analysis will consider the number and degree of exceedance of SQGs.
- The analysis will consider other LOEs such as transition zone water (TZW) compared to ambient water quality criteria for the protection of aquatic life and benthic tissue TRVs.
- The analysis will consider the presence/absence of nearby sources and examine benthic community structure (e.g., via sediment profile imaging and related information).
- The analysis will consider data quality and data density issues for the SQGs.

The LWG's implementation of these guidelines is what has come to be referred to by EPA and the LWG as the comprehensive benthic approach. Developed by the LWG after receiving the EPA's April 21, 2010, directives and guidelines (EPA 2010), the comprehensive benthic approach was first presented informally to EPA (Eric Blischke and Burt Shephard) by the LWG (John Toll and Jim McKenna) on July 20, 2010, to elicit early feedback. It was formally presented to EPA during the September 29, 2010, LWG Small Technical Group Benthic Toxicity AOPCs Meeting with EPA. Item 11 in Attachment B to the LWG's January 12, 2011, letter to EPA (LWG 2011a) and the attachment to EPA's February 25, 2011, response letter to the LWG (Humphrey 2011), document the decision to proceed using the comprehensive benthic approach to develop benthic areas of concern, later termed comprehensive benthic risk areas.

3.2.2.1 Uncertainty Inherent in Relying on an SQV Mean Quotient, or Individual SQV Quotients, to Judge Protectiveness of Potential Remedies

The BERA's benthic risk conclusions used multiple LOEs, including bioassays, benthic toxicity predictions from sediment chemistry data, and benthic community data from sediment profile imaging (SPI). The benthic risk management recommendations focused on defining comprehensive benthic risk areas (see Appendix P, Attachment 1) rather than COCs because the relationship between benthic toxicity and sediment chemistry is strictly correlative and does not identify the cause of toxicity.

Sediment toxicity bioassays form the primary LOE for the comprehensive benthic approach, as per EPA's April 21, 2010, guidelines (EPA 2010). To the extent that the sediment chemistry LOE was used, BERA conclusions focused on most likely level 3 classifications of benthic toxicity (i.e., predictions of toxicity based on more than one model or areas with a high magnitude of toxicity).

Sediment chemistry is a secondary LOE in EPA's guidelines for the comprehensive benthic approach. A mean quotient (MQ) based on the SQVs from the floating percentile model (FPM) (Windward 2011) was one of the tools used to assess the sediment chemistry LOE. An MQ threshold of 0.7 was selected in consultation with EPA. The LWG had originally selected a slightly higher threshold (0.87) based on an analysis of the

MQs at "no hit" bioassay stations, but a value of 0.7 was suggested by EPA and accepted by the LWG. An MQ of 0.7 was selected by EPA in the Problem Formulation based on the value used in the Grand Calumet sediment injury determination (USFWS 2000) and is a conservative assessment of potential future recovery of benthic SMAs given that, in the absence of bioassay data, an area with an MQ of 0.7 would currently be assumed to likely have acceptable levels of benthic risk as is discussed further in Section 5.4. The other tool used to assess sediment chemistry is the maximum probability of toxicity (pMax) from EPA's logistic regression model (LRM)[15]. The calculated pMax was compared to pMax thresholds (0.50 and 0.59) that EPA selected based on an analysis of LRM false positive and false negative prediction rates.

The decision to use these particular MQ and pMax thresholds to assess sediment chemistry was a risk management decision. Sensitivity of SMA boundaries to the selected thresholds is low because bioassays were the primary LOE for this analysis.

Potential future conditions cannot be assessed with bioassays, so sediment chemistry became the primary LOE in the analysis of draft FS alternatives. The MQ threshold was used for that purpose, so the MQ threshold of 0.7 is effectively an RG. Following are some of the major considerations with using the MQ threshold as an RG:

- SQVs are used to calculate the MQ. The models that were used to develop the SQVs do not provide unique solutions to the predictions of toxicity, in part, because the relationship between sediment contaminant concentrations and toxicity is based on correlation, not causation, and the overall incidence of toxicity in Portland Harbor is low. Different FPM runs can result in slightly different groups of contaminants predicting toxicity (although contaminants that predict the majority of the toxicity tend to be included in all models). Selection of contaminant models for both the FPM and LRM relies on best professional judgment for a number of steps, which also contributes to slightly different outcomes in different model runs.
- The effect of non-CERCLA contaminants and other stressors that will potentially not be addressed by remedial action is also an uncertainty in SQV development. Several of these non-CERCLA stressors were included in some of the models (total petroleum hydrocarbons, ammonia, and sulfides in bulk sediment) but were not carried forward as SQVs.[16] Also, several of the individual contaminant that went into EPA's LRM confound CERCLA contaminants with organic enrichment and physical characteristics of the sediment because the LRMs used the COC concentration is normalized to organic carbon. It is unknown to what extent these non-CERCLA stressors contributed to observed toxicity and thus affected the models used to predict toxicity.

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[15] The LRM predicts toxicity based on the correlation between a chemical and toxicity. As with the FPM, the LRM cannot establish causation.

[16] SQVs were derived for these chemicals and a discussion of their potential contribution to benthic invertebrate risk was provided in the risk characterization and uncertainty sections of the draft final BERA.

- As mentioned previously, the selection of the MQ threshold was a risk management decision. Alternative values could have been selected. Since bioassays were the primary LOE for defining comprehensive benthic risk boundaries, the MQ assumption was of secondary importance in that decision. It is more important for analyzing draft FS alternatives because for that we do not have bioassay data. For example, consider two alternatives for an SMA. If it could be predicted, sediment COC concentrations in some specified number of years post-remediation could result in MQs of 0.65 and 0.75. These MQs fall on opposite sides of the MQ threshold, but fundamentally, there may be no measurable difference between the two alternatives in terms of potential benthic community risk. The MQ threshold is a protectiveness criterion, but it is a weak measure of protectiveness. Balancing criteria should weigh heavily in the alternatives analysis and uncertainty about predicted sediment concentrations, SQVs, pMax values, and MQs should be carefully considered. Later on, perhaps at the pre-design stage, additional bioassays and sediment chemistry data, and perhaps benthic community data, should be considered in cases where the benthic analysis is influencing remedial decisions.

3.2.2.2 Uncertainty about Recovery between Now and When Remedies are Implemented

An important assumption in the analysis of alternatives is the rate of recovery of contaminated sediments and TZW from the time that potential risks to the benthic community were characterized to completion of remediation. Therefore:

- Analyses of remedy effectiveness should reflect the requirement that all source controls, including TZW, be in place before sediment remedies are implemented. The overall effect is that source controls would reduce potential baseline risk. As an example, additional upland groundwater source controls would serve to strand the toes of contaminated groundwater plumes, thus reducing groundwater flux to the river, accelerating recovery times, or both.
- Analyses of remedy effectiveness also should account for the lag time between BERA data collection and remediation. This duration may be similar to the period needed for the sediments to naturally recover to concentrations that no longer pose potential risk to the benthic community, particularly for small risk areas associated with only a few COCs or at lower magnitude.

3.2.3 Recommendations for RGs for Ecological Receptors

The sensitivity analysis found that total PCB RGs up to at least 200 μg/kg satisfy the CERCLA protectiveness criterion for ecological receptors in the Portland Harbor Site. PCB RGs greater than 1,000 μg/kg satisfy the CERCLA protectiveness criterion for ecological receptors, except river otter.[17]

[17] The river otter PRG could in the future be found to be overly protective because the current assessment uses mink as a surrogate for characterizing PCB effects. This is conservative because mink is known to be highly sensitive to

While the PCB RGs for mink are protective of the benthic community, other aspects of potential risk need to be considered in thinking about benthic RG and SMA sensitivity. These other aspects include:

- The comprehensive benthic risk areas (see Appendix P, Attachment 1) are defined using a risk management tool that was developed through a long, collaborative process between the LWG and EPA, based on a set of guidelines provided by EPA (2010). This means that , as a matter of a risk management decision, there should be a relatively high level of certainty about benthic-related SMA boundaries, at least as a process matter for the draft FS.
- The tools available for evaluating alternatives' protectiveness of the benthic community are relatively imprecise. Therefore, balancing criteria should weigh heavily in the alternatives analysis, and uncertainty about predicted sediment concentrations should be carefully considered.
- If protectiveness of the benthic community is influencing remedial action decisions in a particular SMA, additional data may be warranted to inform design decisions.

3.3 HUMAN HEALTH RGs

The overall objective of the BHHRA (Kennedy/Jenks 2011) was to evaluate whether exposure to contaminants in sediment, surface water, groundwater seeps, or biota may result in potentially unacceptable risks to human health. The BHHRA was conducted in accordance with technical guidance and other requirements set forth by EPA and incorporated multiple assumptions in estimating the potential risks. Per agreement with EPA, contaminants were identified as potentially posing unacceptable risks if they resulted in a cancer risk greater than 1×10^{-6} or a hazard quotient (HQ) greater than 1 under any of the BHHRA exposure scenarios, which encompass ranges of exposure assumptions at multiple exposure areas with ranges of exposure point concentrations, regardless of the uncertainties. This analysis focused on COCs.

As described above, in April 2009, prior to the completion of the BHHRA, EPA identified Focused PRGs for protection of human health. The Focused PRGs were identified for specific contaminants, exposure pathways, and assumptions from the BHHRA. The exposure scenarios that were the basis of the Focused PRGs for protection of human health are: non-Tribal adult consumption of smallmouth bass for total PCBs, adult consumption of clams for BaP, and Tribal fisher direct contact with in-water sediment for carcinogenic polycyclic aromatic hydrocarbons (cPAHs). As described in draft FS Section 3, for draft FS purposes the smallmouth bass PCB and sediment direct contact BaP goals are recommended as RGs, while the clam BaP goal is recommended as a PRG due to the overall uncertainties associated with this exposure scenario. The

PCB reproductive effects, but the data that would be needed to revise the river otter effects characterization do not currently exist.

Focused PRGs that EPA identified in 2009 are now referred to as EPA's point estimates within a range of potential RGs (PRGs in the case of the clam scenario) for these exposure scenarios.

The sensitivity analysis examined the assumptions in the exposure scenarios that were the basis of the EPA point estimate RGs to identify the range of RGs that would be protective of human health. The technical details of the sensitivity analysis are included in Attachment 2.

3.3.1 RG Sensitivity Analysis Approach

The sensitivity analysis evaluated the impacts of the assumptions used in the BHHRA on the human health RGs using the risk equations from the BHHRA. Probabilistic techniques were used to examine how the use of various exposure assumptions could result in ranges of RGs that would be considered protective of human health. Distributions were applied to the exposure assumptions used in the BHHRA based on the same sources of exposure data that were used in the BHHRA. Monte Carlo simulations were then used to evaluate the cumulative effects on risk estimates of the possible range in values for each of the different exposure parameters.

The approach of using Monte Carlo simulations in an iterative fashion to derive RGs is consistent with EPA guidance (2001). This approach involves determining the exposure point concentration (EPC) that results in risk levels at or below acceptable levels for a given probability of occurrence based on the parameter distributions for the exposure assumptions (such as the 95th percentile for reasonable maximum exposure (RME) scenarios). The forward-facing risk equations from the BHHRA were run a number of times (iteratively) using progressively lower values for the concentration term until the acceptable risk levels were achieved. The successive mock EPC values were then plotted with the corresponding risk estimate, and the best-fit line was used to determine the EPCs corresponding to specific risk levels.

In the sensitivity analysis, EPC values were varied in the Monte Carlo simulations to determine the relationship between the EPC and the risk estimates. For direct contact scenarios, the EPC is equal to the sediment RG. Thus, the relationship developed between the EPC and risk results provides an understanding of the sensitivity of the RGs to different target risk levels and probability percentiles. For bioaccumulation scenarios, the EPC is the target tissue level, which was used in existing sediment-tissue relationship models to calculate the sediment RG. The food web bioaccumulation model was used to develop the sediment RGs for PCBs based on target tissue levels for smallmouth bass.

3.3.2 RG Sensitivity Analysis Results

Through the sensitivity analysis, a range of RGs were developed for PCBs for adult consumption of smallmouth bass by varying BHHRA exposure parameters in a manner consistent with probabilistic risk analysis techniques (see Attachment 2). These parameters are summarized in Table 3.3-1 though additional detail is provided in Attachment 2.[18]

Table 3.3-1. Parameters Evaluated in Human Health Sensitivity Analysis

BHHRA Sensitivity Analysis	Parameter Name	Distribution
Adult consumption of smallmouth bass	Exposure Duration (years)	Lognormal
	Tissue Type Preference (fraction of diet)	Discrete Uniform
	Cooking Preference (fraction of diet)	Discrete Uniform
	Cooking Loss (fraction)	Cooking method specific - Determined based on best fit of tabulated values
	Ingestion Rate (grams/day)	Gamma based on best fit of summary statistics
	Body Weight (kg)	Lognormal
Tribal fisher direct contact with in-water sediment	Exposure Frequency (days/year)	Triangular
	Exposure Duration (years)	Triangular
	Sediment Ingestion Rate (mg/day)	Triangular
	Skin Surface Area (cm^2)	Normal
	Adherence Factor (mg-cm^2)/event	Lognormal
	Body Weight (kg)	Lognormal

The range of RGs reflects different target risk levels and different percentiles from the Monte Carlo simulations. In addition, RGs were developed separately for consumption of whole body tissue, fillet tissue with skin, and fillet tissue without skin using the mechanistic bioaccumulation model (Windward 2009b). The impacts of mechanistic bioaccumulation model uncertainty were also assessed for total PCB RGs protective of smallmouth bass consumers, in the same way the bioaccumulation model uncertainties were assessed for mink in the BERA. PRGs for smallmouth bass consumption were developed for diets consisting only of whole body fish. The extent to which smallmouth bass is consumed as whole body tissue versus fillet tissue versus skinned fillet tissue is not known; however, these latter fillet tissue RGs were calculated for this sensitivity analysis. RGs were developed for cancer and noncancer endpoints. The RGs represent the highest sediment concentrations that would not result in potentially unacceptable risks greater than the target risk levels for the given probability percentile based on the human health sensitivity analyses. The uncertainty associated with the bioaccumulation model discussed in Section 3.1 was used to provide ranges of confidence on the RGs resulting from the human health sensitivity analyses.

[18] It should be noted that there is uncertainty in the assumed distributions of the exposure parameters in addition to other inputs to the probabilistic analyses. This uncertainty was not evaluated in this sensitivity analysis.

Total PCB RGs resulting from the sensitivity analyses for specific target cancer and noncancer levels are presented in Table 3.3-2. The RG ranges due to uncertainty in the bioaccumulation models are presented in the parentheses following the primary estimates of the RGs for the specified target tissue level. For example, Table 3.3-2 shows that at the 90th percentile for a 1 x 10^{-4} cancer risk, the RG for PCBs in skinless fillet tissue of smallmouth bass ranges from 4,470 µg/kg to 8,240 µg/kg. The ranges of RGs for cancer and noncancer endpoints are also shown in Figures 3.3-1 and 3.3-2, respectively. The results of the sensitivity analysis demonstrate that a range of RGs would be protective of human health for purposes of fish consumption and that many of these RGs are higher than the Focused PRG for PCBs of 29.5 µg/kg for smallmouth bass consumption. In addition, it is important to consider the benefits of fish consumption relative to potentially unacceptable risks from fish consumption, as the probability of benefits is likely higher than the potentially unacceptable risks of cancer (Stone and Hope 2010).

A range of RGs for cPAHs that would be protective of direct contact with in-water sediment were also developed as part of the sensitivity analysis in a manner consistent with probabilistic risk analysis techniques (see Attachment 2). The RGs for cPAHs were developed on the basis of BaP toxicity equivalence (BaPEq). Because this exposure scenario is based on direct contact, a bioaccumulation model was not needed to calculate RGs, and the range of RGs resulting from the analysis reflects uncertainties associated with the risk model, but not the bioaccumulation model. The BaPEq RGs resulting from the sensitivity analyses for specific target cancer risk levels are presented in Table 3.3-3. Table 3.3-3 shows that an RG of 1,437 µg/kg would achieve the 99th percentile for a 1 x 10^{-6} cancer risk to Tribal fishers from direct contact with cPAHs in in-water sediment. The RGs developed in the sensitivity analysis for this scenario that are protective of human health range from 1,437 µg/kg to 370,198 µg/kg (90th percentile 1X10^{-4}cancer risk). These RGs are all higher than EPA's preferred point estimate RG for cPAHs of 423 µg/kg for direct sediment contact by a Tribal fisher, which represents a greater than 99th percentile estimate. The upper range for the 1X10^{-6} cancer risk is 3,702 µg/kg, which is still above EPA's preferred point estimate RG for cPAHs of 423 µg/kg for direct sediment contact by a Tribal fisher.

The sensitivity analysis also examined the assumptions used in the BHHRA for exposure to BaP from clam consumption. Though a range of clam tissue levels were calculated that would be considered protective of human health by varying the exposure parameters used in the BHHRA in a manner consistent with probabilistic risk analysis techniques (see Attachment 2) RGs were not developed as discussed further below. As with the sensitivity analysis for PCBs and smallmouth bass consumption, most of the clam tissue levels resulting from the sensitivity analysis were higher than the tissue level that was used to derive the EPA's preferred point estimate PRG. However, the sensitivity analysis also considered the uncertainties associated with the clam consumption scenario. For example, the extent to which clam consumption occurs at the Site is unknown. Also, the areas in which clam consumption would occur are limited to shallow beach areas where harvesting is possible, so the draft FS only considers this scenario in limited areas. Finally, the relationship between BaP in sediment and clam tissue is weak (r^2=0.36)

(Windward 2009b). For these reasons, goals could not be developed for clam consumption with the same certainty as the other scenarios. As described in draft FS Section 3, these goals are referred to as PRGs due to this additional uncertainty. In addition, the RGs for direct contact with in-water sediment are considered protective of potential human exposures to BaP, and thus PRGs for clam consumption are not presented in this document. RGs for direct contact with BaP in in-water sediment are presented in Table 3.3-3.

Table 3.3-2. RGs for PCBs for Smallmouth Bass Consumption

Applicable to 1-RM spatial scales

Target Tissue Level (µg/kg)	Cancer Risk			Noncancer Hazard			Whole Body Sediment RG (µg/kg)	Fillet with Skin Sediment RG (µg/kg)	Fillet without Skin Sediment RG (µg/kg)
	90%	95%	99%	90%	95%	99%			
2			$1x10^{-6}$				--	--	--
15		$1x10^{-6}$					--	3 (≤5)	18 (10-24)
20						1	--	6 (1-9)	26 (15-34)
35			$1x10^{-5}$				--	14 (7-20)	49 (31-64)
41	$1x10^{-6}$						--	18 (9-24)	58 (38-76)
91					1		2 (≤3)	45 (29-60)	135 (92-176)
137		$1x10^{-5}$					5 (≤8)	71 (47-93)	205 (141-267)
246				1			13 (6-18)	131 (89-171)	373 (259-485)
372			$1x10^{-4}$				23 (13-30)	201 (138-261)	566 (396-735)
414	$1x10^{-5}$						26 (15-34)	224 (155-291)	630 (441-819)
470 (EPA's point estimate of the RG)							29.5		
1,356		$1x10^{-4}$					95 (64-124)	744 (521-967)	2075 (1459-2695)
4,140	$1x10^{-4}$						300 (208-390)	2282 (1605-2963)	6346 (4470-8240)

Notes:
Percentiles shown at the top of each cancer risk and noncancer hazard column indicate percentiles of the risk distribution output associated with the target tissue level.
Target tissue levels are in micrograms per kilogram (µg/kg) on a wet weight basis.
Sediment RGs are in µg/kg on a DW basis.
"--" indicates that the target risk level cannot be met at any sediment RG due to the water contributions of PCBs.
RGs are presented as best estimates with uncertainty range (in parentheses) based on bioaccumulation model uncertainty.

Table 3.3-3. RGs for BaPEq for In-Water Sediment Direct Contact

Sediment RG (µg/kg)	Cancer Risk		
	90%	95%	99%
423 (EPA point estimate of the RG)			
1,437			$1x10^{-6}$
2,750		$1x10^{-6}$	
3,702	$1x10^{-6}$		
14,367			$1x10^{-6}$
27,496		$1x10^{-5}$	
37,020	$1x10^{-5}$		
143,673			$1x10^{-4}$
274,960		$1x10^{-4}$	
370,198	$1x10^{-4}$		

Notes:
Sediment remediation goals (RGs) are in micrograms per kilogram (µg/kg) on a dry weight basis.
Percentiles shown at the top of each cancer risk column indicate percentiles of the risk distribution output.
EPA's point estimate of the RG results in a cancer risk of 1×10^{-6} at greater than the 99th percentile.

4.0 BACKGROUND SENSITIVITY

Just as there are sensitivity ranges in the RGs, there are similar sensitivities in the methods and assumptions used in the determination of 'background.' This section summarizes the analysis of sensitivity associated with background estimates for total PCBs (combined)[19] that can be compared to ranges of RGs and RALs. The typical purpose of such comparisons is to identify RGs that are below background. EPA has indicated that, on this project, RGs below background will generally not be used to define SMAs for the draft FS (EPA 2009 and EPA 2005). EPA has focused this determination on one method presented in the draft final RI (described more below) of calculating background based on the upstream bedded sediment dataset. The purpose of this analysis is to explore the sensitivity range of that background estimate using the upstream bedded sediment data and to determine whether additional RGs might reasonably be considered to be at or below background levels. A full description of the background sensitivity analysis and results can be found in Attachment 3.

Several factors contributing to the sensitivity of background values based on the upstream bedded sediment dataset and their comparison to the range of potential RGs have been evaluated. They are:

- Identifying an appropriate background statistic (e.g., upper prediction limit [UPL] and upper tolerance limit [UTL]) for comparison
- The basis (DW or organic carbon [OC]-normalized) for comparison to Site data
- Selection of outliers
- Reliance on a point estimate approach rather than a population comparison approach (i.e., hypothesis testing)
- The handling of data below the detection limit for calculating total or summed values (i.e., total PCBs)
- Spatial autocorrelation among background data that may reduce the effective sample size of the dataset (resulting in increased uncertainty in background statistics).

4.1 CHOICE AND APPLICATION OF BACKGROUND STATISTIC

EPA has in the past focused on the use of a background UPL calculated using DW total PCB concentrations. To evaluate the sensitivity range associated with this choice of statistic (UPL) and basis (DW), several other relevant approaches to defining a background threshold value (BTV; i.e., not-to-exceed value) were considered. As described in the ProUCL guidance (EPA 2007), BTVs can be used to evaluate the effectiveness of remediation through comparison with Site data to determine if remediated Site concentrations are comparable to background level concentrations.

[19] SMA mapping analysis indicates that this contaminant is of greatest interest relative to potential impact of background concentrations on RG selection as well as SMA development.

The choice of statistic to represent the BTV was evaluated by: 1) evaluating alternative approaches to calculating the UPL; and 2) considering the following alternative BTVs discussed in the ProUCL guidance (EPA 2007):

- Non-parametric UPL based on Chebyshev Inequality
- Non-parametric UPL based on ordered statistics
- UPL of mean
- UTL
- Central tendency estimates (i.e., upper confidence limit [UCL])
- p*100 Percentile (i.e., 95th percentile)

In addition, sensitivities in the basis of the sediment concentration were evaluated by considering data on both a DW and OC-normalized basis.

4.1.1 Application of a UCL95 as a BTV

Though UCLs are generally used to calculate EPCs and not BTVs, this statistic was added to the evaluation to provide an analysis of the uncertainty associated with central tendency estimates of background data. Estimates of the mean and associated confidence limits (i.e., upper confidence limit of the mean) were calculated and included in the range of alternative point estimate background statistics) for comparison purposes based on a previous regional application. This approach is consistent with how ranges of BTVs were calculated for the Lower Duwamish River.

However, any application of this statistic in comparison to a Site central tendency estimate (i.e., SWAC) is not considered a statistically robust approach and should be performed with caution. The UCL95 of the background data provides an upper bound estimate of the mean with known level of confidence. It does not provide any known level of confidence when compared to another statistic. That is, the confidence in any conclusions based on a comparison of a background UCL and, for example, a Site SWAC between two sets of data (i.e. Site data greater than background) is unknown. In contrast, a population comparison approach, UPL of the mean, or UTL provides a specified level of confidence (e.g., 95 percent) when testing hypotheses about whether Site and background data are significantly different or not.

4.2 SELECTION OF OUTLIERS

In Section 7 of the draft final RI (Integral et al. 2011), outliers were initially chosen using graphical and statistical analysis (e.g., potential outliers). These outliers were further investigated using Site history information to identify outliers associated with known proximal sources or using other LOEs (e.g., outlier over mean ratio).

To evaluate the full range of sensitivities in the selection of outliers, BTVs were calculated with no outliers removed. In addition, BTVs were calculated using only those primary outliers identified based on their proximity to a known source; primary outliers that could not be tied to a known source were not removed.

4.3 HYPOTHESIS TESTING APPROACH

An alternative approach for larger datasets (n>10) is to apply a hypothesis testing approach. In this approach, statistical tests are performed to test hypotheses regarding the mean, median, or fixed proportion of the background relative to a fixed threshold (i.e., PRG) and to ultimately make some judgment as to whether the RG being considered is significantly less than the background dataset.

In this approach, if the RG is determined to be significantly less than the selected background statistic (e.g., mean), this would be evidence that the RG is below background and should not be used to establish remedial goals. As an example, hypothesis testing was conducted using the background dataset for total PCBs (combined) from Section 7 of the draft final RI and the lower than 5th percentile confidence interval sediment point estimate PRG for mink (31 μg/kg) discussed in Section 3.2.

4.4 SUBSTITUTION METHODS AND NON-DETECT HANDLING

For the draft final RI, contaminant concentrations for multiple-constituent analytical totals (e.g., total PCBs) were calculated using four rules established for the baseline risk assessments. These rules included:

1. Non-detect (ND) values that exceeded the maximum detect for that analyte were excluded from the analytical totals.
2. Contaminants that were never detected in a given background dataset were excluded from the multiple-constituent analytical totals.
3. NDs were included at one-half of the reporting limit for those analytes that were detected at least once in the background dataset.
4. If all analytes contributing to a sum were not detected in a given sample, then the highest reporting limit for any of the individual constituents within the given sample was reported for the total and qualified with a non-detect flag (i.e., U-qualified).

To assess the sensitivities associated with these rules, each rule was varied individually according to the chart in Figure 4.7-1, total values recalculated, and summary BTVs were generated. Finally, all rules were varied to 'bookend' the range of potential summing scenarios. Details of each rule are provided in Attachment 3.

In addition, total PCBs were calculated following the example of Helsel (2010), in which the Kaplan-Meier (KM) mean of individual PCB Aroclors or congeners was calculated on a per sample basis in the background dataset. This mean was then converted to a total PCB value by multiplying by the number of Aroclors or congeners in that sample.

4.5 SPATIAL AUTOCORRELATION

Positive spatial autocorrelation, which is the lack of independence among observations due to their neighboring physical locations, was assessed in the background dataset. Positive spatial autocorrelation can result in an 'effective sample size' that is less than the number of observations in the sample, resulting in an increase in the uncertainty in background statistics (e.g., UPL) calculated using such a dataset. Spatial autocorrelation was assessed using Moran's I statistic (Moran 1950) calculated using ArcGIS. In lieu of more advanced statistical methods for adjusting for spatial autocorrelation, the *ad hoc* method of Dale and Fortin (2002) was used to adjust the significance level and recalculate BTVs if the presence of significant autocorrelation in the background dataset was indicated.

4.6 GEOSTATISTICAL EVALUATION OF BACKGROUND

SWACs were estimated using the bedded sediment background dataset in order to allow a more one-to-one comparison with Site-wide SWACs[20]. Several methods of interpolation were considered; kriging, natural neighbors (NN), and Thiessen polygons. Due to the distribution and other confounding factors (i.e., spatial autocorrelation), the dataset was not amenable to kriging. The other two approaches (NN and Thiessen polygons) are functionally equivalent though mechanically different. On a DW basis, estimates of total PCB background SWACs are 5.01 µg/kg and 5.53 µg/kg using Thiessen polygons and NN, respectively. SWACs based on OC-normalized data are 575 and 596 µg/kg-OC using Thiessen polygons and NN, respectively. Thus, the two different interpolation approaches yield similar SWAC estimates, illustrating that the propagation uncertainty is low.

Estimates of variability between the observed and predicted sediment concentrations (i.e., predictive ability) for the NN SWACs[21] were estimated using the coefficient of variation (CV) of the root mean square error CV(RMSE), which estimates the variability between the observed and predicted values (i.e., RMSE) normalized to the average observed concentration and expressed as a percentage. For the DW SWAC estimated using NN, the CV(RMSE) is 21 percent while the OC-normalized SWAC CV(RMSE) is 11 percent; indicating that the OC-normalized SWAC is a better estimator of background than the DW SWAC.

Although there is good agreement between interpolation methods and the predictive ability of the NN interpolation method was fair, caution should be used in selecting a background SWAC as an RG for the same reasons as stated in Section 4.1.1. Application of a background SWAC in comparison to a Site SWAC is not considered statistically robust. The SWAC does not provide any known level of confidence when compared to another statistic (i.e., Site SWAC). That is, the confidence in any conclusions based on a

[20] SWAC estimates were performed using the SW and OC-normalized datasets used in Section 7 of the draft final RI.

[21] Estimates of variability between observed and predicted sediment concentrations for Thiessen polygons were not available.

comparison of the Site and background SWAC (i.e., Site greater than background) is unknown.

4.7 SUMMARY

Results of each sensitivity analysis are presented in detail in Attachment 3 and are summarized in Figure 4.7-2, 4.7-3, and 4.7-4. In summary, using the various methods described above, the sensitivity for the upstream bedded sediment background value ranges from approximately 5 µg/kg to 37 µg/kg DW, as compared to EPA's chosen statistic of 17 µg/kg DW. (This represents the range of outcomes of calculations described in Sections 4.1 through 4.5.) The upper end of this range is more than two times higher than EPA's chosen statistic. The overall conclusion of this analysis is that significant range of potential background values can be calculated using reasonable and scientifically valid concepts that are not recognized by EPA in their selection of a single background statistic (i.e., UPL DW) for all uses of background in the draft FS. Therefore, RG ranges should be compared to the range of background values resulting from the sensitivity analysis summarized in this section and detailed in Attachment 3. Such an approach is entirely consistent with EPA's current framework where one set RG value is compared to one set background value, but in this case, ranges of RGs and background values should be compared, as discussed further in Section 6.

4.8 MEASUREMENT UNCERTAINTY

The assessment of uncertainty in RGs and background estimates focuses on uncertainties from exposure assumptions and statistical methods. There is the additional uncertainty related to measuring low levels of COCs that is important in evaluating achievement of RGs, particularly RGs near or within the range of background estimates. The sources of measurement uncertainties include known acceptable levels established prior to a sampling event (i.e., analytical precision or accuracy identified in a Quality Assurance Project Plan [QAPP]) and relatively unknown sources (e.g., ability in the field to accurately sample an intended area of sediment consistently). Outside of any uncertainties in the calculation of an RG, measurement uncertainties may limit the ability to refine SMAs in remedial design that are intended to achieve very low RGs and accurately determine whether an RG has been achieved after a remedy is completed.

For example, a relative percent difference (RPD) of 50 percent is established for the LWG QAPP as a conservative target control limit for variations in concentrations between field duplicates and/or split samples for results detected at greater than 5 times the reporting limit (which is approximately 5 ppt for individual PCB congeners). Given that the RG ranges discussed here are well above 5 times the reporting limit, this RPD limit of 50 percent generally applies. Therefore, for sampling attempting to test compliance with the smallmouth bass whole body EPA point estimate PRG of 29.5 µg/kg, the concentrations in field duplicates could acceptably vary from 14.75 to 44.25 µg/kg. This acceptable measurement range extends from well below the PCB background Focused PRG of 17 µg/kg to above the LWG high estimate background value of 37 µg/kg and nearly equal to the smallmouth bass fillet with skin consumption non-cancer RG of 45 µg/kg. Thus, differences in only the most disparate of the RG estimates presented in this uncertainty analysis could be routinely identified in any future monitoring program using best available sampling and measurement techniques. This issue is even further compounded when evaluating the PCB congener method, as the summed non-detects (or half detection limits) themselves may be in this range of acceptable measurement uncertainty.

5.0 SMA SENSITIVITY

One key use of sediment RGs is to define SMAs, which is accomplished through SMA mapping procedures. These procedures contain assumptions that can be changed to other scientifically valid assumptions, creating a range of sizes and shapes of the SMAs that can be used in the draft FS. The remainder of this section analyzes various sensitivities associated with these SMA mapping procedures.

5.1 REVIEW OF MAPPING PROCEDURES AND RALS

Before exploring variations on the procedures to map SMAs, it is necessary to describe these general procedures. The SMA definition procedures include the following primary elements, in approximately step-wise order:

1. **Define Dataset** – The existing surface sediment chemistry dataset is defined using a consistent set of rules, particularly for handling NDs and summing of NDs for summed contaminants (e.g., total PCBs). The draft FS dataset has been defined using data quality and data reduction rules agreed to between LWG and EPA, which include using RI data quality rules and the risk assessment summing rules. In particular, these rules include using half the detection limit for undetected contaminants and summing of NDs for summed contaminants.
2. **Apply Data Spatially** – Existing surface sediment chemistry data are applied to the Site area on a consistent horizontal spatial basis such as using Thiessen polygons or various contouring algorithms. For this project, NN contouring is the standard method agreed to with EPA.
3. **Map SMAs Using RALs** – Areas that would need to be remediated (i.e., SMAs) to meet RGs over an exposure area consistent with the risk assessments are identified and mapped on the spatial basis established. As discussed more in Section 4 of the draft FS, this mapping is achieved through RALs. RALs developed using the QEAFATE chemical fate model predictions of future post-remediation conditions that vary over time. The fate modeling approach allows an assessment of RALs that can achieve various RGs at specified time periods following remediation (incorporating all of the fate and transport processes represented in the model), and simultaneously assesses long-term chemical changes over the entire Site including areas that are not dredged or capped.
4. **Identify Mapping Artifacts** – The SMAs defined are further evaluated to handle some obvious artifacts that can be created by the mapping process. One of these artifacts is that the contouring algorithm may identify areas of elevated concentrations that are unreasonably disassociated with contaminant data locations. This can occur in situations where the program does not readily "recognize" physical barriers and boundaries of the Site, or some spatially isolated stations represent very large areas merely due to a lack of data in these areas. Another artifact is that SMAs may be mapped for RGs representing risk scenarios in areas where no such risks were actually found in the risk assessment.

This occurred for mapping of BaPEq RALs as described in draft FS Section 5. This by itself is an important form of uncertainty in the RGs related to the extrapolation from the risk assessment data (e.g., risks estimated from tissue data) to sediment data (e.g., through the bioaccumulation model or other biota-sediment relationships). Although a relationship between sediment and tissue data, for example, can be found, this relationship has some level of error that creates uncertainties in the application of RALs to meet RGs.

The range of SMA outcomes created by some of the procedures in this overall SMA mapping process is discussed in Sections 5.2 through 5.5.

In addition to these procedural steps, there is another large variation in SMAs associated with the selection of RALs to map such that certain RGs within the overall range of RGs described in Section 3 are met. This is described in Section 5.6. Although this is fundamentally a risk management decision, which is discussed more in Section 6, understanding the relationship between RALs and the attainment of RGs over time is critical to developing a rationale for such risk management decisions and understanding the context of those decisions.

Finally, note that benthic SMAs are not mapped using RALs or the above approaches. Multiple lines of bioassay, contaminant concentration, and other evidence are used to define benthic SMAs via the comprehensive benthic approach as described in Section 3.2.2.

5.2 NON-DETECT HANDLING ANALYSIS

The assumed value of NDs included in sums for classes of contaminants (e.g., total PCBs) can substantially affect the concentration contours used in SMA mapping, which in turn impacts the SMA sizes. The impact of these ND assumptions on SMA mapping was quantified using total PCBs as the most relevant example, as it influences the majority of SMA size.

In the standard LWG RI/FS calculation of total PCBs for a particular sample, if all PCB congeners in that sample are ND, then the highest detection limit is used as the total PCB concentration and the sample is qualified ND. If there is at least one detected congener, results for each individual congener are included in the total PCB sum at one half of the detection limit, and the sample is qualified as a detected sample, which could result in relatively high ND total PCB sums when summing detection limits in samples with a high percentage of non-detected congeners.

To quantify the uncertainty associated with NDs, example total PCB RALs of 1,000 µg/kg and 75 µg/kg were mapped using three assumptions for NDs in the summing process:[22]

[22] High non-detects (defined as non-detect results 25 times above detection limits) were not included in the dataset used to generate NN contour surfaces).

- Zero for NDs
- Full detection limit for NDs
- Historical project approach of assuming one half of the detection limit for NDs.

It is recognized that both the values of zero and full detection limit are unlikely to be accurate across the entire dataset. If the actual concentrations below detection limits were known, they would be expected to be most often somewhere between these bounds. However, the use of zero for NDs in sums is not unprecedented and was used, for example, in the Duwamish FS (AECOM 2010).

Figure 5.2-1 (RAL of 1,000 µg/kg) and Figure 5.2-2 (RAL of 75 µg/kg) depict the SMA mapping results using this range of assumptions for total PCB calculations. Note that "U=1" in the figure legends indicate the assumption of full detection limits for NDs. The differences in SMA sizes are larger for mapping of lower RALs under the various ND assumptions as shown in Figure 5.2-2. A summary of the difference in SMA areas (in acres) for each of the ND handling procedures is summarized in Figure 5.2-3 for five alternatives evaluated in the draft FS (see draft FS Sections 5 and 7 for descriptions of each alternative in detail). This figure confirms that the uncertainty in the SMA size increases as the RAL mapped decreases, and thus, the uncertainties are greatest for the mapping of the largest overall SMAs (at a RAL of 75 µg/kg overall SMA acreage differed between 209 and 337 acres). The uncertainties caused by the handling of non-detects in generating concentration contours are considered moderate as compared to other SMA uncertainties discussed here.

5.3 DATA DENSITY AND NATURAL NEIGHBORS CONTOURING ANALYSIS

Sensitivities associated with handling of data density artifacts and NN contouring procedures were also evaluated. As noted above, data density mapping artifacts can be created when one or a few stations represent concentrations in the contouring process for a very large area for a certain COC. Because the draft FS data are biased towards shoreline areas, these data density issues most often occur in the navigation channel. Additional data acquisition could be considered during pre-remedial engineering design studies to further define the aerial and vertical extent of these COCs prior to engineering design. When such an isolated station is above the RAL being used, this can result in a very large portion of an SMA being identified. Although it is possible to simply use this very large area for the purposes of the draft FS, in such situations it is much more likely that additional samples would be collected nearby during remedial design and reduce the actual size of the SMA. Consequently, the SMA mapping procedures used to date on the project have applied a "data density" buffer to recognize this more likely outcome and produce a more technically valid SMA area for draft FS purposes. These buffers are often only in the navigation channel where data density is often less than compared to shoreline/nearshore areas. The buffer is essentially a circle of defined diameter placed around the station that sets the limit of area that the station will represent in the mapping and can vary by COC. Figure 5.3-1 shows various data density buffers that could be applied using the example of mapping a total PCB RAL of 75 µg/kg and standard NN contouring. This example shows buffers applied in the navigation channel only as follows:

- No buffers applied. The NN contouring is unrefined.
- The buffer distance is based on the average distance between stations in the navigation channel (206 feet). In this case, only stations in the navigation channel that are more sparsely spread than the average density in the channel are buffered. This is the approach applied in the draft FS.
- The buffer distance is based on the average distance between stations in the entire Site (179 feet). Only stations in the navigation channel that are more sparsely spread than the average density for the entire Site are buffered.

As shown in Figure 5.3-1, the buffering assumption makes a moderate difference in SMA size for a few select areas in the navigation channel (overall SMA acreage differed between 285 and 311 acres).

5.3.1 Natural Neighbors Contouring

Additionally, variations in SMA mapping caused by use of one particular contouring approach (i.e., NN contouring) was evaluated by comparing the contoured surfaces generated with natural log-transformation of the data prior to NN contouring with surfaces generated with no transformation. This is a standard feature in ARC-GIS and is often used to reduce the influence of extreme and isolated values on the resulting contours. Figure 5.3-2 shows an example of the SMA differences that are created, again using the example of Alternative F using a total PCB RAL of 75 µg/kg. As shown in Figure 5.3-2, the SMAs created by a different contouring assumption can be noticeably different in some areas. The use of log NN contouring creates smaller SMAs where data density is lower, includingAOPCs 23, 14, 16, and 18. The resulting decrease in overall SMA size across the entire Site using the log NN approach is approximately 30 percent.

It should be noted that a wider range of uncertainty exists in SMA mapping procedures than is discussed here based on the contouring method selected that was not quantitatively assessed as part of the sensitivity analysis. All of the uncertainty evaluations regarding SMA mapping conducted in this appendix use NN contouring as the contouring method, though there are other types of contouring methods (e.g., Kriging and inverse distance weighting [IDW]) that could be used to generate SMAs. Although not quantitatively assessed here, it is likely that differences in SMAs sizes using other contouring methods would be at least as large, and perhaps larger, than the overall SMA size variations discussed above.

5.3.2 SMA-Specific Data Density Issues

The sensitivity to uncertainties in the handling of data density artifacts is an issue that can and will likely affect remedial design on an SMA-specific basis. The shape and size of an SMA is directly the result of the density (number and location) of data within the SMA itself. Thus, assumptions in addressing these data density issues within an SMA may have more of an influence in SMA-specific remedial design than in an evaluation of alternatives for the Site as a whole. For example, data density assumptions used in the handling of select dioxin/furan results along the boundaries of SMA 13 (Willamette Cove) affect the overall size of the SMA as there are only a few samples in the vicinity and thus, could impact the remedial design for that SMA. Since the buffer distance used in the draft FS was chosen based on a Site-wide basis, there are likely cases where another buffer distance or another approach in delineating a SMA is more applicable. This aspect of uncertainty in data density assumptions was evaluated to introduce potential issues that might arise in remedial design of specific SMAs through applying an alternative approach to SMA development using dioxin/furan sediment data and the EPA-directed PCDF RAL of 0.02 µg/kg (which is a surrogate for dioxin/furan compounds), focusing on SMA 13. Further detail on the development of the dioxin/furan RALs is presented in Section 4 of the draft FS.

Figure 5.3-3 shows that limited data density is influencing SMA size under the methods used in the draft FS where there is only one surface location (containing three samples) with PCDF concentrations (which is used as a surrogate for dioxin/furans) above the EPA-directed RAL of 0.02 µg/kg (06R002). Given the few results and that only one location exceeds the RAL used in the in this example, it is unlikely that when evaluated on an SMA-specific basis, specific cleanup actions would be conducted based on this existing data density.

Figure 5.3-3 provides an example of one alternative SMA shape that could be developed if either alternative assumptions were made about data extrapolation and/or additional sampling was conducted to further delineate PCDF towards the boundaries of the SMA shape presented in the draft FS. The SMA shape is shown as would be defined in the draft FS using only the PCDF RAL of 0.02 µg/kg. Additionally, for this example, an alternative SMA shape is shown by applying an approximate buffer distance to all samples in this SMA by assuming half the distance between 06R002 (the only location which exceeds the example RAL in this SMA) and the nearest existing sample locations below any PCDF RALs for GWC1 and BT016 and then extrapolating out the same distance on the other side of 06R002. Essentially, this disregards the existing potentially outlying surface points of G665 and C291 in defining the SMA size and shape and instead uses the other closer points (to the 06R002) to make assumptions about what might be the case if other nearby surface data existed and results were similar to GWC1 and BT016.

This example illustrates one type of future remedial design uncertainty where extrapolation between data points within an SMA with limited data may cause large differences in SMA size, and therefore large differences in cleanup volumes and cost, especially if there are only a few data points with results above RALs surrounded by results below RALs (as in the case of the SMA 13 example). There may be reason to use a different approach in extrapolating data than that used in the draft FS based on individual SMA characteristics. This issue becomes more influential on an SMA-specific basis especially for contaminants influencing SMA size that are surrogates for other chemical totals (i.e., PCDF for dioxins/furans for the SMA 13 example) where variability in concentration gradients between the surrogate data and the data for which the surrogate is being used might differ within an SMA as compared to assumptions made on a Site-wide basis.

5.4 BENTHIC SMAS

As noted in Section 3.2.2, the natural recovery (or attenuation) of contaminant concentrations over time should be considered for benthic SMAs, but methods to accomplish this are not readily available. To illustrate an example of the uncertainty that may be involved in assessing natural recovery of benthic areas over time, the contaminant concentration data within the comprehensive benthic approach SMAs was examined.

This evaluation focused on MQ values. (The pMax values were also briefly evaluated, but it was found that these were highly variable both inside and outside the benthic SMAs and were not further investigated.) Within the benthic SMAs, 64 percent of the stations have MQ values below 0.7 and 44 percent have MQ values below 0.3. Given the uncertainties in the derivation of the MQ values and the somewhat limited correlation between this measure and bioassay toxicity results, the large number of low MQs within benthic SMAs is not particularly surprising. However, bioassay results themselves cannot easily be used to predict future conditions at the Site after any natural recovery takes place. To the extent that MQs are likely the best measure of the relationship between chemistry and bioassay results, MQs can be used as at least an approximate measure of areas that might naturally recover in the future. On this basis, areas with benthic SMAs that are currently below an MQ of 0.7 were excluded from the benthic SMAs, and the remaining areas were mapped as shown in Figure 5.4-1, which also compares these reduced areas to original benthic SMAs. The difference between these reduced areas and the original areas is moderate (64 percent difference in overall acres). As stated in Section 3.2, an MQ of 0.7 was selected by EPA in the Problem Formulation based on the value used in the Grand Calumet sediment injury determination (USFWS 2000) and is a conservative assessment of potential future recovery of benthic SMAs given that, in the absence of bioassay data, an area with an MQ of 0.7 would currently be assumed to likely have acceptable levels of benthic risk.

Figure 5.4-1 can only be considered an approximate estimate of areas that might have the potential to naturally recover for benthic toxicity. In that context, this evaluation suggests that there are considerable portions of the current benthic SMAs that might be considered suitable candidates for either:

1. Additional bioassay testing to confirm or refute toxicity as a part of remedial design, or
2. As part of an overall monitoring program for a natural recovery, with appropriate contingency measures should these areas not show actual recovery over a reasonable time period (e.g., 5 to 10 years).

5.5 UNCERTAINTY OF MAPPING BaPEq

The range of RGs for cancer risk for Tribal fisher direct contact with sediments is based on total cPAHs, expressed as a BaPEq concentration. However, the fate and transport model evaluates BaP as a single contaminant. The toxicity of the same concentration of BaP or BaPEq is essentially equivalent. Thus, any differences in the application of these RGs is related to the variations in concentrations of the single contaminant BaP versus the concentration of all cPAHs including BaP (expressed on a BaPEq basis) present in the same sediment sample.

BaPEq is calculated by multiplying the seven typically analyzed cPAHs by their respective potency equivalent factors (PEFs), and summing the resulting concentrations. PAHs classified as carcinogenic are benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, BaP, indeno(1,2,3,-c,d)pyrene, and dibenzo(a,h)anthracene.

PEFs were assigned according to EPA (1993):

Analyte	PEF
Benzo(a)anthracene	0.1
Benzo(a)pyrene	1
Benzo(b)fluoranthene	0.1
Benzo(k)fluoranthene	0.01
Chrysene	0.001
Dibenzo(a,h)anthracene	1
Indeno(1,2,3-cd)pyrene	0.1

The difference between mapping BaP and BaPEq is shown in Figure 5.5-1 using RALs of 1,500 µg/kg and 20,000 µg/kg. This mapping reveals that as the RAL increases, the variability in the sizes of SMAs mapped decreases, similar to the effect of non-detect handling discussed above. This relationship is shown more completely in Figure 5.5-2 (the difference in overall acreage from the current FS approach ranges between approximately 10 and 30 percent between the various RALs). Given that RALs used in the draft FS are greater than 1,000 µg/kg (except for Alternative G, which was screened out in draft FS Section 7), the overall change in SMA size created by modeling and mapping on a BaP basis is relatively minor.

5.6 MAPPING RANGES OF RGs VIA RALs

So far three types of general sensitivities have been explored in this analysis:

- The sensitivities associated with developing an RG based on the risk assessment and bioaccumulation models (Section 3)
- The sensitivities associated with background estimates to which the RGs can be compared (Section 4)
- The sensitivities associated with mapping those RGs using RALs (Sections 5.2 through 5.5).

This section examines the uncertainty in the development of the RALs and in the sensitivities in the relationship between the RALs and RGs that those RALs can be assumed to attain.

Tables 5.6-1 and 5.6-2 present the select RGs discussed in Section 3 on a consistent scale for total PCBs and BaPEq, respectively. The RGs are shown on the left side of each table. On the right side of both tables, the RALs are plotted on the same scale, but in this case the RALs are plotted at the Site-wide SWAC that they are calculated to achieve at time zero (immediately after construction), Year 10, and Year 30. These tables allow quick comparison of the various RGs, and the RALs that would be expected to attain them, over various periods of time. The SWACs attained by each RAL are calculated using the "best estimate" from the QEAFATE modeling process described previously. Consequently, these tables do not include the uncertainty in the RALs, which is discussed next.

The RALs that would achieve smallmouth bass RGs on a Site-wide basis are plotted in Table 5.6-1. However, EPA has indicated that smallmouth bass exposures should be evaluated on a river mile basis. Thus, RGs that are achieved on a Site-wide basis may not achieve the same RG on a river mile basis for every river mile. This issue was assessed in draft FS Section 4 and Appendix Db regarding RAL development, and it was found that a given smallmouth bass RG that was met at a specified time using a range of RALs on a Site-wide basis was also generally met on a river mile basis, except for two river miles. For these two river miles, the RG might still be met at a later point in time. As discussed later in this appendix, it is technically unreasonable to treat a particular RG as a specific value with no sensitivity or uncertainty bounds around it. When evaluating attainment of smallmouth bass RGs, it should be considered that the actual exposures are unlikely to be on exactly a river mile basis, the RGs tend to be nearly met (or met in a somewhat longer time period) in the two river miles, and the RGs are usually more than met in the remaining river miles over the same time periods. In the context of the overall range of RG uncertainties, examination of RALs that achieve smallmouth bass RGs on a Site-wide basis as shown in Table 5.6-1 is a technically valid simplifying approximation for the purposes of this uncertainty discussion. The draft FS comparisons of alternatives is conducted on a segment and river mile basis, so this assumption is not used for all purposes in the draft FS (see Section 8 of the draft FS).

Figures 5.6-1, 5.6-2, and 5.6-3 plot the Site-wide SWACs that are achieved by PCB RALs for the same three time periods (zero, 10, and 30 years after construction) as compared to select RGs and the range of background values presented in previous sections. These three figures present similar information but on two different y-axis (SWAC) scales that allow examination of both relatively high RGs (Figure 5.6-1 shows select ecological and human health RGs) that are well above the current Site-wide PCB SWAC (approximately 85 µg/kg) as well as RGs that are within the range of the current Site SWAC (Figure 5.6-2 for ecological RGs and Figure 5.6-3 for human health RGs). Figures 5.6-2 and 5.6-3 also show the uncertainty in the RALs themselves as determined through QEAFATE modeling evaluations. These figures also show the ranges of outcomes using various methods to calculate time zero SWACs, which do not rely on the QEAFATE model, but still require some assumptions about the concentrations present in active remediation areas immediately after construction is completed. Note that the RAL associated with Alternative F (75 µg/kg), does not show up as a distinct data point in these figures; however, the SWACs attained by such a RAL can be estimated using the curve developed based on other RAL points. This is the primary purpose of such RAL curves, so that not all possible RALs have to be modeled in order to make a selection of the particular RALs for use in the draft FS. Additionally, for RGs for smallmouth bass consumption, fillet without skin are not represented on these figures as the range of those RGs is covered on the lower end of the range by the fillet with skin RGs and does not add significant information to the figures.

Figures 5.6-1 through 5.6-3 show that the sensitivity range of the RGs far exceeds the uncertainty in the RALs. The current Site-wide SWAC and any decrease in the SWAC provided by RALs over time, are essentially indistinguishable in Figure 5.6-1 as compared to the range of RGs, indicating that the overall range of RGs consumes the variability in the SWACs resulting from any RAL. Also the uncertainty in the RALs is small when compared to the range of RGs available below the Site SWAC. Further, the SWAC levels that the RALs would be expected to attain, and many of the RGs relevant to these same levels, reside within the range of background values for the Site, indicating that there is a wider range of RAL-based alternatives that may meet background conditions.

Figures 5.6-4 and 5.6-5 show similar information for BaPEq RGs and RALs. In the case of BaPEq, Tribal fisher direct contact with sediment risk scenario is the primary RG evaluated in this sensitivity analysis. Consistent with human health risk assessment exposure assumptions, this RG is applied on a shoreline half-river mile basis. AOPC 9U (Figure 5.6-4) and AOPC 6 (Figure 5.6-5) approximately correspond to two shoreline half river miles where BaPEq concentrations are generally the most elevated for the entire Site. Therefore, the full range of RALs evaluated can be best evaluated by examining these two half river mile examples. The situation for BaPEq is similar to the overall pattern seen for PCBs. Specifically, the half river mile SWACs, and any decrease in those provided by RALs over time, are relatively small as compared to the overall range of the RGs.

Because the RG sensitivity ranges are so large, risk management decisions related to RG selection take on primary importance in determining SMA size. For example, any RGs currently above the Site SWACs will result in no SMAs, and any RGs below background, particularly EPA's definition of background, will essentially identify most, if not all, of the Site as an SMA.

This analysis shows that any RAL that attains levels somewhere slightly below the current Site SWACs and slightly above background (however one chooses to measure that given the uncertainty in background estimates) can be related to achieving some particular RG within this range of various RGs. Thus, the selection of RALs and SMAs becomes primarily a risk management decision about what RGs to focus on within this relatively limited range of concentrations. Figures 5.6-6 through 5.6-10 show examples of various RALs that could be related to RGs within the range of the Site SWACs and EPA's definition of background level. It is noteworthy that many of the figures appear relatively similar. That is, the same RALs can be related to several different types of RGs within this range. The legends of these figures (and reference to Tables 5.6-1 and 5.6-2) are the key to understanding how each SMA developed from each RAL can be related to the RGs within this range. These figures include the following examples of SMA development:

- Figure 5.6-6: Human health smallmouth bass whole body cancer risk at a 10^{-4} risk level for total PCBs using RALs that meet an RG range from the 99^{th} percentile (23 µg/kg to 95^{th} percentile (95 µg/kg), and including EPA's point estimate of the smallmouth bass RG of 29.5 µg/kg, which falls within this range.
- Figure 5.6-7: Human health smallmouth bass fillet with skin noncancer risk for total PCBs using RALs that attain RGs between background and the Site-wide SWAC. This approximately equates to the range of RGs from the 99^{th} percentile (6 µg/kg) to 90^{th} percentile (131 µg/kg) estimates, although the lower end is below background and the upper end is somewhat higher than the current Site SWAC.
- Figure 5.6-8: Ecological mink total PCB risk using the range of exposure and bioaccumulation RG estimates below the existing Site SWAC as shown in Table 5.6-1.
- Figure 5.6-9: The range of technically valid total PCB background estimates from Section 4 with EPA's Focused PRG background value used as the lowest value.
- Figure 5.6-10: Human health Tribal fisher direct contact with sediments BaPEq cancer risks at a 10^{-6} risk level using RALs that meet an RG range from the 95^{th} percentile (2,750 µg/kg) to greater than the 99^{th} percentile (i.e., EPA's point estimate RG of 423 µg/kg).

This analysis demonstrates that even the highest RALs attain a wide range of RGs given the uncertainties associated with the RGs alone, suggesting that balancing criteria, such as short-term effectiveness and cost should be considered in selecting remedies for the Site.

6.0 OVERALL SENSITIVITY AND CONCLUSIONS

This section discusses the overall ranges of sensitivities observed for RGs, background estimates, SMA mapping procedures, and RAL estimates, all of which have an impact on the mapping of SMAs for the draft FS.

6.1 HIERARCHY OF RG/SMA SENSITIVITIES

As discussed in Section 5.6, there is a sensitivity range for RGs that, if mapped via RALs, result in SMAs that range from designating the entire Site as meeting RGs that are protective of health and the environment, to identifying the entire Site as a single continuous SMA. Thus, there is no need for a formal calculation that attempts to propagate all the RG, background estimate, SMA mapping, and RAL estimates sensitivities through to an "overall SMA sensitivity." The RG ranges alone provide the entire possible bounds to any such exercise. Consequently, it is more useful to examine the extent to which each type of sensitivity analyzed can and will contribute to variations in SMA sizes in any given circumstance.

Even though RGs alone can provide the entire range of possible SMA sizes, it is still important to examine the contribution of different sources of uncertainty to SMA size when making risk management decisions. As each source of uncertainty is considered, decisions are often made regarding the need to select a conservative approach for the particular uncertainty being considered without regard to the effect such a series of conservative decisions may have on the outcome. The net result may be to 'compound' uncertainty throughout the analysis of interest to arrive at an estimate or result that does not reflect the initially desired level of 'protectiveness,' but rather, results in an outcome that is highly conservative, which has implications for other factors (e.g., cost and feasibility) in cleanup decisions. Although it is beyond the scope of this current analysis to quantify the effects of every possible combination of conservative assumptions for each type of uncertainty discussed above, the compounding effects of multiple conservative decisions in the face of uncertainty is still important to consider. Even though multiple decisions may not result in an extreme outcome such as designating the entire Site as an SMA, those decisions may result in substantial additions to SMA sizes, and as a result substantially impact the evaluation of alternatives relative to issues of effectiveness, implementability, and cost.

Based on the work presented in the previous sections, the following hierarchy of sensitivities exists:

- **Largest sensitivities:**
  - **RG Sensitivities** – The ranges of sediment RG values extend from zero (due to water column inputs that cause potentially unacceptable risk under some exposure assumptions) to orders of magnitude above the current Site SWACs.
  - **Selection of RGs for SMA Determination (i.e., Risk Management)** – The selection of RGs within the bounds of background and the current SWACs to

determine RALs for SMA mapping creates the next largest range of sensitivities. This is also an EPA risk management decision. Eventually, EPA will need to examine the RG ranges and make a risk management decision about which of these RGs and associated RALs should be used to determine the focus of cleanup efforts at the Site.

- **Medium sensitivities**:
  - **Background Estimate Sensitivities** – The background range falls within the above sensitivity ranges, with reasonable estimates for total PCBs ranging from about 5 μg/kg, the lower bound discussed above, to 37 μg/kg, which is a little less than half the current Site SWAC.
  - **SMA Mapping Sensitivities** – These are the issues discussed in Sections 5.1 through 5.5. Their approximate subhierarchy of sensitivity within this category are likely:
    - NN contouring
    - Benthic SMA, data density buffering, and ND handling
    - BaP vs. BaPEq mapping
  - **RAL Development Uncertainties** – There are substantial uncertainties in defining the RALs, either on a time zero basis or using QEAFATE modeling for year 10 and 30 estimates as presented in Section 5.6.

The SMA uncertainties summarized above can be assessed in a combined fashion that results in a range of SMA sizes that fall within the ultimate bounds of no SMAs on the one side and the entire Site as an SMA on the other side. The SMA uncertainties alone do not represent the complete sensitivity range. However, as noted above regarding the effect of compounding risk management decisions, it is useful to see the sum total impact of the SMA uncertainty analyses including variations in NN contouring, benthic SMAs, data density buffering, and ND handling. Figure 6.1-1 shows the largest calculated SMAs, using RALs of 75 μg/kg for PCBs and 1,500 μg/kg for BapEq, the full comprehensive benthic risk areas, and the following conservative SMA mapping assumptions combined:

- Detection limits in sums are set to the whole detection limit
- Data density buffer is set equal to navigation channel average density
- Non-transformed NN contouring is used

Figure 6.1-2 shows the smallest calculated SMAs for the same RALs, reduced comprehensive benthic risk areas representing some assumed natural recovery in areas below an MQ of 0.7, and using the following least conservative SMA mapping assumptions combined:

- Detection limits in sums are set to zero

- Data density buffer is set equal to Site-wide average density
- Natural log transformed NN contouring is used

Although the range of SMAs across these combined assumptions is smaller than the overall range possible using variations of other possible RGs, the difference between the two SMA variations in Figures 6.1-1 and 6.1-2, just looking at mapping procedure assumptions, is considerable. Approximately a dozen AOPCs that show some type of SMAs mapped within the AOPC boundary black lines in Figure 6.1-1 using the conservative assumptions show relatively insignificant SMAs with respect to impact on overall Study Area acreage (SMAs less than one half acre in size) using the lesser conservative assumptions in Figure 6.1-2 (reducing SMA sizes within AOPCs between 20 percent and 98 percent). As discussed above, the SMA mapping decisions alone can result in substantial effects on SMA size due to multiple compounding decisions, and thus, these decisions should be made as a group, not as individual conservative decisions without regard to the others. However, as discussed in Section 5.3, SMA-specific issues (e.g., data density unique to SMA-specific COCs) might need to be evaluated on and individual SMA basis during remedial design.

6.2 CONCLUSIONS

There is a large sensitivity range associated with the identification and selection of RGs and the resulting SMAs that can be mapped. The conclusions from this analysis are:

1. Overall Sensitivities: The sensitivity ranges for RGs are well beyond the current Site background (lower bound) ranges and the current Site SWACs (upper bound). These ranges ultimately determine the sum total of the range of SMAs that can be developed including no SMAs at all.

2. Ecological RGs and Benthic SMAs:

    a. The sensitivity analysis for mink shows that EPA's point estimate for the mink total PCB RG of 31 μg/kg is lower than necessary to protect the mink population, and that a technically valid RG for mink is significantly higher than EPA's point estimate RG. The mean estimate is 256 μg/kg, which is several times higher than the current Site SWAC of approximately 85 μg/kg.

    b. The mink total PCB RG will be protective of bald eagle and otter given the existing Site SWAC.

    c. Natural recovery of potential benthic risk in benthic SMAs is a significant source of uncertainty that is difficult to quantify. Uncertainty about EPA's position on the draft final benthic BERA and the comprehensive benthic approach is another source of significant uncertainty. Uncertainties also exist about the MQ threshold exceedance approach to assessing remedy effectiveness that cannot be satisfactorily quantified within the working draft FS schedule.

3. Human Health RGs: A range of RGs is protective of human health. For smallmouth bass consumption, most of the PCB RGs that are protective of human health are higher than EPA's point estimate RG of 29.5 µg/kg. For direct contact with in-water sediment, all of the cPAH RGs that are protective of human health are higher than EPA's point estimate RG of 423 µg/kg.
4. Background: Significant sensitivity ranges exist in the calculation of background values (from approximately 5 µg/kg to 37 µg/kg) that is not recognized by EPA in their selection of a single background statistic (i.e., UPL DW of 17 µg/kg) for all uses of background in the draft FS. Many of the RGs within this background range are not consistent with EPA policy to generally not set cleanup levels below background levels.
5. SMA Mapping and RAL Development Uncertainties: Substantial sensitivity ranges exist with various SMA mapping procedures and the calculation of RALs that generally fall within the sensitivity ranges created by the above issues, but can have an important additive impact on the overall SMAs sizes in particular locations. This is particularly true when the discussion is confined to relatively low RGs such as EPA's point estimates of the RGs, where the combined SMA sensitivities can have large impacts on individual SMAs.
6. Selection of RGs and RALs: The sensitivity ranges for RGs extend from below background to above the current SWACs, which emphasizes the importance of risk management decisions selecting RGs and the RALs that meet the RGs over various time periods. These risk management decisions ultimately control the determination of SMA sizes. Eventually, EPA will need to examine the RG uncertainties and make a risk management decision about which of these RGs should be used to determine the focus of cleanup efforts at the Site.

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